FINAL

Smoky Canyon Mine
Remedial Investigation/Feasibility Study

Site-Specific Ecological Risk Assessment Report

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95UCL	95 Percent Upper Confidence Limit
AET	Apparent Effects Threshold
AMSL	Above Mean Sea Level
AWERA	Area-Wide Ecological Risk Assessment
AWQC	Ambient Water Quality Criteria
AWWQRP	Arid West Water Quality Research Project
BLM	Bureau of Land Management
BPF	Baseline Problem Formulation
BW	Body Weight
BW/day	Body Weight per Day
C	Concentration
CaCO3	Calcium Carbonate
CDPHE	Colorado Department of Public Health and Environment
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
cfs	cubic feet per second
CO	Consent Order
COPC	Chemical of Potential Concern
CSM	Conceptual Site Model
DEQ	Department of Environmental Quality
DQO	Data Quality Objective
ECOPC	Ecological Chemical of Potential Concern
EcoSSL	Ecological Soil Screening Level
ECSM	Ecological Conceptual Site Model
EPC	Exposure Point Concentration
ERA	Ecological Risk Assessment
ERED	Environmental Residue Effects Database
ER-L	effects range-low
FETAX	Frog Embryo Teratogenesis Assay Xenopus
HQ	Hazard Quotient
IBI	Index of Biotic Integrity

ACRONYMS (continued)

IDAPA	Idaho Administrative Procedures Act
IDEQ	State of Idaho Department of Environmental Quality
IDFG	Idaho Department of Fish and Game
IFWIS	Idaho Fish and Wildlife Information System
IR	Ingestion Rate
kg	kilograms
kg/day	kilograms per day
kg/kg/day	kilograms per kilogram per day
LANL	Los Alamos National Laboratory
LEL	Lowest Effect Level
LOAEL	Lowest-Observed Adverse Effects Level
LOEs	Lines of Evidence
mg/kg	milligrams per kilogram
mg/kg-BW/day	milligrams per kilogram body weight per day
mg/kg dw	milligrams per kilogram dry weight
mg/L	milligrams per liter
NAWQC	National Ambient Water Quality Criteria
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No Observed Adverse Effects Level
NOEC	No Observed Effects Concentration
NTCRA	Non-Time-Critical Removal Action
ODA	Overburden Disposal Area
PEC	Probable Effects Concentration
PEC-Q	Probable Effects Concentration Quotient
RBT	Rainbow Trout
RI COPC	Remedial Investigation Chemical of Potential Concern
RI/FS	Remedial Investigation/ Feasibility Study
RME	Reasonable Maximum Exposure
SAP	Sampling and Analysis Plan
SEIS	Supplemental Environmental Impact Statement
SI	Site Investigation
SLERA	Screening Level Ecological Risk Assessment

ACRONYMS (continued)

SMI	Stream Macroinvertebrate
SQG	Sediment Quality Guideline
SQUIRTS	Screening Quick Reference Tables
SSERA	Site-Specific Ecological Risk Assessment
SSHHRA	Site-Specific Human Health Risk Assessment
SUA	Special Use Authorization
SUF	Site Use Factor
T/E	Threatened and Endangered
TEC	Threshold Effect Concentration
TRV	Toxicity Reference Value
TSMD	Tri-State Mining District
TTF	Trophic Transfer Factor
UCL	Upper Confidence Limit
UET	Upper Effects Threshold
ug/L	micrograms per liter
USACHPPM	U.S Army Center for Health Promotion and Preventive Medicine
USEPA	United States Environmental Protection Agency
USFS	USDA Forest Service
USFWS	U.S. Fish and Wildlife Service
YCT	Yellowstone Cutthroat Trout

1.0 INTRODUCTION

This Site-Specific Ecological Risk Assessment (SSERA) presents an evaluation of potential risk to ecological receptors at the Smoky Canyon Phosphate Mine (Site) in southeastern Idaho, as required under the Administrative Settlement Agreement and Order on Consent/Consent Order (Settlement Agreement/CO) for Remedial Investigation/Feasibility Study (RI/FS) entered into by Simplot, the USDA Forest Service (USFS), the U.S. Environmental Protection Agency (USEPA), and the State of Idaho Department of Environmental Quality (IDEQ) (USFS, USEPA, and IDEQ 2009). The USFS is the Lead Agency, and the USEPA, the U.S. Department of the Interior Fish and Wildlife Service (USFWS) and Bureau of Land Management (BLM), IDEQ, and the Shoshone-Bannock Tribes (Tribes) have elected to participate as Support Agencies (collectively referred to as the "Agencies").

The RI was initiated for the Site in 2009, and included data collection from spring 2010 through fall/winter 2012/2013. The RI/FS, as specified in the 2009 Settlement Agreement/CO is designed to fulfill the following general objectives:

- Determine the nature and extent of contamination and any threat to the public health, welfare, or the environment caused by the release or threatened release of hazardous substances, pollutants or contaminants at or from the Site, by conducting a remedial investigation; and

- Determine and evaluate alternatives for remedial action to prevent, mitigate or otherwise respond to or remedy any release or threatened release of hazardous substances, pollutants, or contaminants at or from the Site by conducting a feasibility study.

The purpose of this SSERA is to provide an analysis of potential baseline risks and help determine the need for remedial action at the Site, in conjunction with the RI and the FS (Formation Environmental [Formation] 2014). The Final RI Report (Formation 2014) describes the Site background, and sources for 22 chemicals of potential concern (COPCs) that were identified by the USFS for investigation at the Site. The Final RI Report also describes the nature and extent of these RI COPCs in various environmental media, and contaminant fate and transport at the Site.

The Smoky Canyon Phosphate Mine is located on National Forest System land in the Caribou/Targhee National Forest approximately 24 miles directly east of Soda Springs, Idaho and is operated under a USFS Special Use Permit and Bureau of Land Management (BLM) phosphate leases (Figure 1-1). The Site is defined by the 2009 Settlement Agreement/CO as the Smoky Canyon Phosphate Mine and includes: areas of overburden disposal associated with the mine; the areal extent of contamination associated with those features; and suitable areas, in very close proximity to the areal extent of contamination, necessary for response action implementation (Figure 1-2). Specific mining and mine-related areas of the Site include backfilled Panels A, B, C, D, and E (Figure 1-2); the external overburden disposal areas (ODAs) associated with these mine panels; and the Pole Canyon cross-valley fill ODA. The investigation area for the SSERA (referred to as “Site-wide”) is defined as the mine-related areas, as well as the areas downgradient and downstream of the mine-related areas where samples were collected during the RI, including Sage Valley, Crow Creek, Hoopes Springs, Sage Creek, Smoky Creek, South Fork Sage Creek, and Tygee Creek.

1.1 Purpose and Scope

The SSERA determines whether the combination of Site-specific exposure scenarios and measured levels of ecological COPCs (ECOPCs) pose current or future potential risks to ecological receptors, and supports the RI/FS and development of remedial action alternatives. The ECOPCs are identified from the list of 22 RI COPCs identified in Attachment B to the 2009 Settlement Agreement/CO Statement of Work. The methodology used in this SSERA was developed in conjunction with Agency input and was outlined in the Smoky Canyon Mine SSERA Work Plan (Formation 2011a) and subsequent Baseline Problem Formulation (BPF) (Formation 2013).

The SSERA structure follows the USEPA eight-step Ecological Risk Assessment (ERA) process (USEPA 1997). The eight steps are:

- Step 1: Screening-Level Problem Formulation;
- Step 2: Screening-Level Exposure Estimate and Risk Calculation;
- Step 3: Baseline Risk Assessment Problem Formulation;
- Step 4: Study Design and Data Quality Objectives Process;
- Step 5: Field Verification of Sampling Design;
- Step 6: Site Investigation and Analysis Phase;
- Step 7: Risk Characterization; and
- Step 8: Risk Management.

A flow chart depicting the USEPA ERA process is shown in Figure 1-3.

Extensive information on the physical and ecological characteristics of the Site was developed and presented in the Site Investigation (SI) Report (NewFields 2005) which included data on the distribution of a focused list of chemicals. The SI information was adequate to complete Step 1 of the USEPA process, develop an Ecological Conceptual Site Model (ECSM), identify ecological receptors for the SSERA, and design additional data collection to support the SSERA. Subsequently, the SSERA Work Plan (Formation 2011a) was prepared pursuant to the 2009 Settlement Agreement/CO) (USFS, USEPA, and IDEQ 2009) and included Steps 3, 4,

5. Initial elements of Step 6 were also largely completed in the SSERA Work Plan. However, the 2009 Settlement Agreement/CO required a list of COPCs specifically for the RI (RI COPCs; Table 1-1) that included chemicals not assessed in the SI. The substantial additional data collection and analysis necessary to address these requirements was documented in the Final RI Report (Formation 2014). The additional data were required to complete Step 2 (Screening-Level Exposure Estimate and Risk Calculation) and subsequent identification of COPCs for ecological receptors (ECOPCs)

In the BPF (Formation 2013), Step 3 of the USEPA ERA process, the RI data (based on the complete RI database) were screened to identify ECOPCs. Potential risks of adverse effects were then assessed quantitatively and qualitatively in the SSERA Risk Analysis (Step 6) and Risk Characterization (Step 7) provided in this document. The Risk Analysis and Risk Characterization are intended to aid risk managers at the Site to collectively make scientifically-defensible Risk Management decisions (Step 8) as part of the RI/FS process.

1.2 Document Organization

The SSERA is presented in seven sections. Section 2 provides a presentation of the BPF data which includes the Conceptual Site Model (CSM) and the selection of the ECOPCs originally provided in the BPF (Formation 2013). The Risk Analysis and Risk Characterization are provided in Section 3 for Aquatic Receptors and in Section 4 for Terrestrial and Riparian Receptors. The Uncertainty Analysis is presented in Section 5. Conclusions are provided in Section 6 and cited references are listed in Section 7. Appendices are provided at the end of the document, and include Appendix A (Site-Wide Data Summary – Abiotic Media); Appendix B (Tier 1 and Tier 2 Wildlife Exposure Point Concentrations); Appendix C (Detailed Exposure and Hazard Quotient Calculations); Appendix D (Overview of Site-Specific Selenium Threshold Development for Brown Trout); and Appendix E (Simplot Responses to Agency Comments on Revised Draft SSERA Report).

2.0 BASELINE PROBLEM FORMULATION

The BPF (Formation 2013) contains the completion of Step 3 in the USEPA ERA process (see Section 1.1). The BPF provides the results of the Screening Level Ecological Risk Assessment (SLERA) that was conducted using the complete RI database, and identifies the ECOPCs for which risk of adverse effects is assessed quantitatively and qualitatively in this SSERA Report. The BPF submittal (Formation 2013) included a detailed description of the relevant Site features, potential ecological receptors, the identification of ECOPCs, and the identification of the assessment and measurement endpoints for the risk characterization. Because that information is germane to the completion of the SSERA and necessary to fully understand the conclusions, the information provided in the BPF is summarized in the remainder of this section. Additional information on the Site setting, presented in the Final RI Report for the Smoky Canyon Mine (Formation 2014), is also provided.

2.1 Physical Setting and Climate

The Smoky Canyon Mine is located in Caribou County, Idaho, within the Southeastern Idaho Phosphate Mining District. Phosphate ore is extracted from the Phosphoria Formation in a series of open pits referred to as mine panels. Panels A through E are located on the eastern slope of the Webster Range between Smoky Canyon and South Fork Sage Creek (Figure 2-1). Elevations range from 6,500 feet to 8,300 feet above mean sea level (AMSL). The slopes of the Site drain generally eastward, with Site streams flowing into the Salt River which flows to the Snake River. The closest main population center to the mine is the Star Valley community, which includes the town of Afton, Wyoming, approximately 10 miles directly east of the mine. The town of Afton has a population of approximately 1,900 (U.S. Census Bureau 2013). Caribou County has a cool and dry climate, with typical prevailing winds and weather patterns moving from west to east. Annual precipitation is approximately 20 to 35 inches per year. The most abundant precipitation occurs in the spring and early summer months. In the winter months, snowfall averages 100 inches each year, and snow cover typically remains on the ground from November to March or April. Summer temperatures in the region normally range from 44 to 82 degrees Fahrenheit, while winter temperatures typically range from 4 to 28 degrees Fahrenheit (Mariah 1988).

2.2 Land Use

The predominant land uses in the vicinity of the Site are associated with agriculture and natural resources, and include crop production (primarily hay) on private lands along with cattle and sheep ranching on private and public lands. Phosphate mining, while not a dominant land use in terms of acreage, is economically important, accounting for approximately 40 percent of the wages and salaries for Caribou County, Idaho. On National Forest System lands, recreational activities include hunting, fishing, camping, hiking, skiing, and snowmobiling, among others. Recreational use of public lands is another important aspect of the local economy. Additionally, parts of the Site may be used for hunting, fishing, and ceremonial activities consistent with the heritage of the Shoshone-Bannock Tribe.

The areas occupied by the mine panels are located on National Forest System lands. These lands currently have or formerly had phosphate leases, are identified as Category 8.2 in the Revised Forest Service Plan for the Caribou-Targhee National Forest, and have restricted public access due to safety concerns (USFS 2003). Current access to the public is restricted by Simplot in the active and inactive/reclaimed mine areas.

Simplot recently purchased private ranch land (former Peterson Ranch) located in Sage Valley immediately east of the Pole Canyon ODA and Panels A, D, and E. Simplot also owns land north of Sage Valley in the Tygee Creek drainage east of Panels A, B, and C. Other private lands are located in the Crow Creek Valley, south and southeast of the Site, and include several ranches and vacation homes (Figure 2-2).

2.2.1 Current Land Use

The mill and administrative and maintenance facilities for the Smoky Canyon Mine are located in Smoky Canyon near the northern end of mining operations (Figure 1-2). Mine Panel A is located immediately east of the mill, Panels B and C are north of the mill, and Panels D and E are south of the mill. The tailings ponds, which are not included within the Site subject to this RI/FS, are located about 3.2 miles northeast of the mill site in the Tygee Creek drainage. The tailings property encompasses 1,680 acres of private land owned by Simplot. The mill is connected to the tailings ponds with a pipeline that extends down Smoky Canyon.

The mining and milling operations for Panels A through E are contained within 2,600 acres of federal phosphate mineral leases administered by the Pocatello Field Office of the BLM and approximately 1,200 acres of Special Use Authorization (SUA) land administered by the Caribou-Targhee National Forest, Soda Springs Ranger District. The mining operations are located on Federal Phosphate Lease Nos. I-012890, I-026843, I-027801, I-27512, and I-30369. Overall, the Panels A through E area extends for a distance of approximately 5.9 miles north to south along the east flank of the Webster Range.

Federal grazing allotments are located on National Forest System lands and overlap with the mine area (Figure 2-2). In general, livestock are not allowed to graze on active mining areas, or on reclaimed or non-reclaimed mine overburden areas. Cattle and sheep graze on the allotments adjacent to the mine areas during the summer months. The private lands in Sage Valley have been used for cattle grazing in the past, but not in the last several years. No permanent or seasonal residences have been utilized in Sage Valley since Simplot acquired the property in 2011. Ranch headquarters are located downstream of Sage Valley on Crow Creek.

2.2.2 Potential Future Land Use

The anticipated future (post mining) land use at the Smoky Canyon Mine includes wildlife use, human recreational activities, grazing, and Tribal activities consistent with their heritage. The majority of the Site is on National Forest System lands in the Caribou-Targhee National Forest, and future recreational use is likely to include activities such as camping and hiking. Future operation and maintenance activities will be conducted at the Site after mining has been completed. In addition, Simplot plans to return grazing as a land use in Sage Valley in the future. Potential future human land uses at the Site are discussed in detail in the Site-Specific Human Health Risk Assessment (SSHHRA) Report (Formation 2015b). Risks to human health are not discussed in this document.

2.3 Reclamation History

Mining activities have been ongoing at the Site since 1983. The original Surface Mine and Reclamation Plan (Simplot 1981) called for mining in five “panels” referred to as Panels A through E. Mining began in Panel A and, subsequently proceeded through Panels D, E, C, and B. To date, all five of the panels have been mined and at least partially reclaimed. Figure 2-3 provides a status map of mine-disturbed and reclaimed areas at the Site based on 2012 data. The timing and status of reclamation are summarized in the following subsections. A more detailed description is provided in Section 5 of the Final RI Report (Formation 2014).

2.3.1 Panel A and Associated ODA

Panel A covers approximately 244 acres, with external ODAs covering approximately 135 acres. Pits A-1, A-2, and A-3 were backfilled around the time of mining in this area. Pit A-3 is partially regraded and will be completely backfilled and reclaimed as part of ongoing mining operations in Panel B. Simplot will reclaim the remaining areas when mining is complete.

2.3.2 Panel B and Associated ODA

As of fall 2012, Panel B covered approximately 270 acres, with additional external ODAs covering approximately 111 acres. Simplot plans to continue mining in Pits B-2 and B-4. Reclamation is planned as these pits are backfilled and graded over the next several years. Reclamation activities such as sloping cover material placement, and/or seeding of the external ODA and the lower portion of Pit B-1 have been completed, although final determination of the completion of reclamation has not yet been made.

2.3.3 Panel C

Panel C covers approximately 104 acres and was mined from 2002 through 2006. Panel C is backfilled with overburden from Panels B and C. The northern end of Panel C (approximately 47 acres) was reclaimed in 2008 and the remainder of Panel C was reclaimed in 2010.

2.3.4 Panel D and Associated ODA

Panel D covers approximately 268 acres with an additional external ODA covering approximately 89 acres. Simplot mined at Panel D from 1993 through 1998. Three pits (Pits D-1, D-2, and D-3) were mined and then backfilled and reclaimed. Panel D overburden was placed in an external ODA as well as in portions of the adjacent Pole Canyon ODA. Reclamation activities such as sloping cover material placement, and/or seeding of the backfilled pits and external ODA were completed in 2002, although the final determination of the completion of reclamation has not yet been made.

2.3.5 Panel E and Associated ODA

Panel E covers approximately 392 acres with an additional external ODA covering approximately 122 acres. Simplot initiated mining in Panel E in 1998 and mining continued through 2006. There are a total of five pits (Pits E-0, E-1n, E-1s, E-2, and E-3) and one external ODA. Mining in Pit E-1n started in 1998 and reclamation of this pit and the external ODA was completed in 2003. Mining began in the other pits in Panel E in 2000 and 2001 and was completed in 2006; the pits remained open or were partially backfilled after mining. Backfilling of Pits E-2 and E-3 began in 2003, and reclamation was completed in 2008. Backfilling and reclamation of Pit E-1s was completed in 2008. Backfilling of Pit E-0 with overburden from other active mining began in 2010 and was completed in 2013. The reclamation activities discussed above included sloping cover material placement, and/or seeding. Note that the final determination of the completion of reclamation has not yet been made.

2.3.6 Pole Canyon ODA

The Pole Canyon ODA is an external disposal area that covers approximately 120 acres in the lower drainage of Pole Canyon Creek just upstream of where the creek enters Sage Valley. The Pole Canyon ODA was constructed in lower Pole Canyon as a cross-valley fill and occupies the Pole Canyon Creek valley between Panels A and D. Most of the overburden in the Pole Canyon ODA originated from Panel A mining between 1985 and 1990. A much smaller portion of the overburden originated from Panel D (Pit D-2) and was placed on the west side of the Pole Canyon ODA in 1997.

The western (upstream) portion of the ODA extends approximately 150 feet above the Pole Canyon Creek bed while the eastern (downstream) portion rises approximately 500 feet above the creek bed. The disposal area extends approximately 4,600 feet from the west face to the eastern toe. The disposal area width (north to south direction) ranges from approximately 1,000 to 1,700 feet. Based on a comparison of pre-mining to current topography, it is estimated that approximately 26 million cubic yards of material are present in the ODA.

Initially, only chert material was placed in the ODA to limit the amount of fine-grained material that entered the Pole Canyon Creek flow. Natural sorting of materials by gravity due to end-dumping resulted in predominantly coarse materials filling the narrow canyon bottom first and the creation of a zone of higher hydraulic conductivity, referred to as a “French drain," through which Pole Canyon Creek flowed prior to diversion in 2007. The plateau and steeper east face of the ODA were reclaimed in 1989 and 1990, and the west face of the ODA was reclaimed in the late 1990s.

In 2006, the USFS selected a Non-Time-Critical Removal Action (NTCRA) to address the transport of selenium and other overburden constituents from the Pole Canyon ODA to surface water and groundwater. Simplot entered into an Administrative Settlement Agreement and Order on Consent/Consent Order for Non-Time-Critical Removal Action for the Smoky Canyon Phosphate Mine in October 2006 (USFS, USEPA, and IDEQ 2006). Details of the NTCRA are provided in the Final RI Report (Formation 2014). Implementation of an additional NTCRA in 2015 is focused on installation of an engineered Dinwoody cover on the ODA.

2.4 Ecological Setting

The ecological/biological setting of the Site was generally characterized through the baseline studies conducted for the Panels B and C Supplemental Environmental Impact Statement (SEIS) (BLM and USFS 2002) and the characterization was originally presented in the SI Report (NewFields 2005). Extensive data regarding the vegetation communities present across the Site were subsequently collected during the RI and presented in the Final RI Report (Formation 2014). Information from these sources is used to describe the ecological and biological setting and conditions at the Site in this section.

2.4.1 Vegetation Communities

Seven general vegetation or habitat types have been identified within and around the Site (Maxim 2002): aspen, conifer, aspen/conifer, mixed shrub, sagebrush, disturbed, and riparian/wetland as shown in Figure 2-3.

Higher elevation areas and north and west aspects at the Site receive sufficient moisture to support species, such as subalpine fir *(Abies lasiocarpa)* and Engelmann spruce *(Picea engelmannii),* with an understory component consisting of pinegrass (*Calamagrostis rubescens*), elk sedge (*Carex geyeri*), and wild strawberry (*Fragaria virginiana*).

Mid-elevation areas at the Site are represented by Douglas-fir (*Pseudotsuga menziesii*) and aspen (*Populus tremuloides*) with an understory of sticky geranium (*Geranium viscossimum*), silver lupine (*Lupinus argenteus*), mountain snowberry (*Symphoricarpos oreophilus*), Indian paintbrush (*Castilleja miniata*), and Kentucky bluegrass (*Poa pratensis*).

Forest openings are dominated by a mixed shrub component that includes species such as mountain snowberry and antelope bitterbrush (*Purshia tridentata*) with an understory consisting of Wyeth buckwheat (*Erigonum herecleoides*), heart-leaf arnica (*Arnica cordifolia*), capitate sandwort (*Arenaria congesta*), bluebunch wheatgrass (*Agropyron spicatum*), and Idaho fescue (*Festuca idahoensis*).

The warmer and drier lower elevation areas and south aspects are typified by mixed shrub communities such as mountain big sagebrush (*Artemisia tridentata*) and grassland species such as bluebunch wheatgrass, Kentucky bluegrass, and western needlegrass (*Stipa occidentalis*). Forbs commonly found in this cover type include silky lupine (*Lupinus sericeus*) and arrowleaf balsamroot (*Balsamorhiza sagittata*).

Riparian areas are dominated by willows (*Salix spp*.), Nebraska sedge (*Carex nebrascensis*), aquatic sedge (*Carex aquatilis*), beaked sedge (*Carex utriculata*), and bluejoint reedgrass (*Calamagrostis canadensis*). Due to extensive grazing in portions of the Site, riparian areas for several of the area streams have been denuded resulting in bank failures and sediment loading.

Selenium accumulation into plants is of particular interest at the Site. Some plant species, including species in the genera *Aster* and *Astragulus,* can accumulate selenium to levels that are acutely toxic to grazing livestock. Concentrations in some of these species can be greater than 1,000 mg/kg (Mackowiak and Amacher 2010). Many other plant species, including common grasses and forb species, cannot tolerate high selenium levels and do not accumulate selenium to high levels.

Based on surveys conducted during site characterization for the RI, selenium hyperaccumulator (e.g., *Astragulus*) and accumulator (e.g., *Aster*) plant species are absent from much of the site which may be due in part to an herbicide program at the Site. None of the species listed by Mackowiak and Amacher (2010) as hyperaccumulators (plants accumulating selenium at concentrations greater than 500 mg/kg) or accumulators (plants accumulating selenium at concentrations from 50 to 100 mg/kg) were observed in any of the transect locations nor in any of the composite samples collected for the RI (Formation 2014) as shown in Tables 2-1 through

2-3. However, alfalfa (*Medicago sativa*) is common in some of the reclaimed areas of the Site and elsewhere in the region, and is considered a selenium accumulator in some investigations.

Some hyperaccumulator and accumulator plant species (Mackowiak and Amacher 2010) were identified during studies conducted prior to the RI. These species were identified in samples collected for the SI in 2003 and 2004 (Appendix B, NewFields 2005). *Astragulus spp.* (milk vetch) was observed at a total of 10 locations in transects on the Pole Canyon ODA, Panel D backfilled pit/ODA, and at seeps DS-7 and ES-4. *Aster spp.* (aster) was observed at a total of 17 locations. Of those, 9 locations were in transects on the Pole Canyon ODA, upland of the Pole Canyon ODA, on the Panel D ODA, and at seep DS-7, and 8 locations were in transects in Sage Valley. Also, weedy milk vetch (*Astragulus miser var. miser*), field milk vetch (*Astragulus agrestis*) and several species of aster (Eaton's aster [*Aster eatonii*], Engelmann's aster [*Aster engelmannii*], leafy aster [*Aster foliaceous*], and few-flowered aster [*Aster modestus*]) were identified in the baseline study for vegetation resources, conducted in 2000 to support preparation of the SEIS for Panels B and C (Appendix A, Maxim 2002).

2.4.2 Vegetation Communities on Panels A, D, E, and Pole Canyon ODA

Vegetation cover and community information was recorded in 2004 for the SI and in July 2010 for the RI at ODA sampling areas to describe current reclamation conditions and to provide additional information for evaluating reclamation alternatives. Vegetation community metrics, such as percent cover, dominant species, and species diversity, were evaluated using quantitative measurements along five transects on each of the seven ODA sampling areas identified for the RI:

- Panel A Area 1;
- Panel A Area 2;
- Pole Canyon ODA;
- Panel D North;
- Panel D South;
- Panel E Area 1; and
- Panel E Area 2.

Given the amount of time between community studies, only the 2010 data are discussed here. A summary of species present and identified in each of the transects is presented on Table 2-1. A total of 69 species were identified in the transect sampling.

In addition, ten randomly selected locations were identified within each ODA sampling area for sampling of soil and biota. At each of these locations (Figure 2-4) quantitative data regarding vegetation community cover information were collected. Table 2-2 summarizes the dominant species that were present in each general sampling area. A total of 34 species were identified at the soil/biota sampling locations.

2.4.3 Vegetation Communities Observed in Riparian Areas, Seeps, and Northern Sage Valley

Vegetation community data were also collected as part of the soil and biota sampling completed in 2010 for the riparian areas, seeps, and northern Sage Valley. Vegetation species composition data were collected in the sampling areas at the following locations (Figure 2-5):

- Smoky Creek;
- Pole Canyon Creek;
- Sage Creek;
- Hoopes Spring (three locations);
- South Fork Sage Creek;
- Seeps DS-7, ES-3, and ES-4; and
- Northern Sage Valley (ten locations).

The dominant species identified at these sampling locations are summarized in Table 2-3. A total of 36 species were identified during the 2010 sampling event.

2.4.4 Wildlife

Wildlife species at the Site are typical for the habitats in the region (Table 2-4). Mammal species include bats, lagomorphs (rabbits), rodents, carnivores, and ungulates. Rodent species that may be found in the area include: meadow vole (*Microtus pennsylvanicus*), long-tailed vole (*Microtus longicaudus*), southern red-backed vole (*Clethrionomys gapperi*), montane vole (*Microtus montanus*), deer mouse (*Peromyscus maniculatus*), chipmunk (*Tamias spp*.), yellow-bellied marmot (*Marmota flaviventris*), porcupine (*Erithizon dorsatum*), and northern flying squirrel (*Glaucomys abrinus*). Lagomorphs are primarily represented by Nuttall's cottontail (*Sylvilagus nuttalli*) and jackrabbit (*Lepus spp*). Documented or suspected carnivores in the area include black bear (*Ursus americanus*), mountain lion (*Felis concolor*), bobcat (*Lynx rufus*), striped skunk (*Mephitis mephitis*), red fox (*Vulpes vulpes*), coyote (*Canis latrans*), badger (*Taxidea taxus*), marten (*Martes americana*), long-tailed weasel (*Mustela frenata*), and ermine (*Mustela erminea*). Ungulates frequenting the area, primarily from spring through fall, include mule deer (*Odocoileus hemionus*), elk (*Cervus elaphus*), and moose (*Alces alces*).

Several species of birds occur in the area, including: raptors, upland gamebirds, passerines, waterfowl, and shorebirds. Raptors that may use the general area for hunting and/or nesting include: Bald Eagle (*Haliaeetus leucocephalus*), Golden Eagle (*Aquila chrysaetos*), Red-tailed

Hawk (*Buteo jamaicensis*), Swainson's Hawk (*Buteo swainsonii*), Northern Goshawk (*Accipiter gentiles*), Cooper's Hawk (*Accipiter cooperii*), Northern Harrier (*Circus cyaneus*), American Kestrel (*Falco sparverius*), Boreal Owl (*Aegolius funereus*), Great Horned Owl (*Bubo virginianus*), and Great Gray Owl (*Strix nebulosa*). With the exception of northern harriers, these raptor species may be expected to nest in aspen or conifer stands. Northern harriers prefer to nest and hunt in grassland habitat near meadows and marshes. Aquatic species such as Mallard Duck (*Anas platyrhynchos*) and Belted Kingfisher (*Megaceryle alcyon*) also inhabit the riparian habitats in the vicinity of the Site.

Game birds commonly found in the area are Greater Sage Grouse (*Centrocercus urophasianus*), Blue Grouse (*Dendragapus obscurus*), and Ruffed Grouse (*Bonasa umbellus*). Sage Grouse preferentially utilize sagebrush vegetation while Blue Grouse and Ruffed Grouse typically are found in dense conifer and aspen stands.

Among the additional bird species that utilize the mine area at times during the year are Hairy Woodpecker (*Picoides villosus*), a year-round resident of coniferous and deciduous forests, American Robin (*Turdus migratorius*), Tree Swallow (*Tachycineta bicolor*), Western Wood-pewee (*Contopus sordidulus*), House Wren (*Troglodytes aedon*), Song Sparrow (*Melospiza meoldia*), Gray-headed Junco (*Junco hyemalis*), and Chipping Sparrow (*Spizella arborea*).

2.4.5 Aquatic Setting

The Site spans a number of local drainages that flow along the eastern portion of the north-south trending Webster Range into the Tygee Creek and Sage Creek basins (Figure 2-6). The Tygee Creek basin drains north to Stump Creek and includes Smoky Creek, Roberts Creek, East Tygee Creek, and Tygee Creek. The Sage Creek basin drains south to Crow Creek and includes Pole Canyon Creek, North Fork Sage Creek, Sage Creek, and South Fork Sage Creek. Both the Tygee Creek basin (by way of Stump Creek) and the Sage Creek basin (by way of Crow Creek) discharge into the Salt River, and the Salt River flows north into the Snake River.

The mine disturbance area and nearby Sage Valley to the east contain several perennial streams and two large springs: Hoopes Spring and South Fork Sage Creek Springs. Average daily high and low flows in the Hoopes Spring channel are 8.36 and 6.34 cubic feet per second (cfs), respectively. Flows have been observed to be nearly constant over years of monitoring due to discharge from the spring complex. Downstream of the South Fork Sage Creek Springs, average daily high and low flows are 10.7 and 8.47 cfs, respectively, which has been relatively constant as well. Unnamed springs with lower flows are found in other areas of the Site.

Historical mining operations have taken place within both the Sage Creek and Tygee Creek drainage basins. Panels D and E, Pole Canyon ODA, and a portion of Panel A are located in the Sage Creek basin. Panels B and C and a portion of Panel A are located in the Tygee Creek basin. Basin characteristics and characteristics of sub-basins within the Tygee Creek and Sage

Creek basins are listed in Table 2-5 along with a complete description of the drainages covered under the RI/FS. Surface water and sediment sampling locations are shown in Figure 2-7.

Tailings Ponds 1 and 2 are located along Tygee Creek upstream of its confluence with Smoky Creek. The tailings impoundments are located outside of Area A (i.e., outside the RI/FS investigation area) but are briefly described here for completeness. The hydrology of the tailings impoundments is primarily controlled by inflow and outflow associated with the mill tailings deposition and reclaim water circuit. The majority of Roberts Creek, upper Tygee Creek and East Tygee Creek flows are diverted around the tailings impoundments in the Roberts Creek Diversion Ditch. Detailed discussions of the tailings impoundment area are presented in the *Draft Smoky Canyon Mine Area B (Tailings Impoundments) Groundwater and Environmental Media Investigation Report* (MFG 2003).

The water resources of the Tygee Creek and Sage Creek drainage basins are primarily used for agricultural applications. Significant portions of Pole Canyon Creek and Sage Creek were used to irrigate private agricultural lands in Sage Valley during the spring and summer. Land in this valley has since been acquired by Simplot and irrigation still occurs, but not to the extent historically. Springs present along the east and west sides of Sage Valley are used for stock watering. In addition to these agricultural uses, a portion of Roberts Creek is used by Simplot to provide additional mill make-up water (BLM and USFS 2002).

The creeks near the Site do not have any special state or federal designations that significantly restrict their use. The USFS has determined that none of the creeks in the Tygee Creek and Sage Creek drainage basins near the Site are eligible for designation as Wild and Scenic Rivers (USFS 1998). Based on the most current State of Idaho 303(d) list of impaired waters cited in the State Integrated Report, North Fork Sage Creek, Pole Canyon Creek, South Fork Sage Creek, and Sage Creek downstream of North Fork Sage Creek are listed as impaired due to selenium (IDEQ 2014). Smoky Creek, Tygee Creek, Roberts Creek, Crow Creek, Sage Creek, and South Fork Sage Creek are listed for non-contaminant impairments such as bacteria, sedimentation, and/or habitat issues.

The creeks within the Tygee Creek and Sage Creek basins are subject to IDEQ water quality standards for their designated uses. All surface waters draining from the Site are designated for cold-water biota use. Water quality conditions in these basins are generally characterized by moderate hardness, low concentrations of suspended solids, and circumneutral pH conditions.

In general, stream flows are low and the creeks do not transport large quantities of sediment except during spring-runoff conditions when creeks may become more turbid. Sediment conditions are generally characteristic of headwater creeks with benthic substrates ranging from near bedrock to sand and cobbles covered by small boulders. Many creeks have notable amounts of fine particles, which result in moderate to high embeddedness of cobbles and small boulders. Fine sediment loads have historically been due to grazing activities in these watersheds where livestock trample banks and denude riparian vegetation. Recent steps to mitigate these effects have been undertaken by Simplot, USFS, and private landowners by fencing off stream areas from livestock use, and resulting improvements in stream bank stability have been noted through continuous monitoring. Mining operations do not generally affect sediment conditions because sediment catch basins and erosion fences are utilized to inhibit off-Site migration of particulates.

Pole Canyon Creek is the only creek that may potentially contain sediments from mined areas, because the creek historically flowed along the base of the Pole Canyon ODA (prior to implementation of the 2006 NTCRA as discussed in Section 2.3.6); slope failure at the ODA toe in the mid-1990s resulted in transport of some overburden to sediments immediately downstream from the ODA (Formation 2014). Flows from upper Pole Canyon Creek are now diverted via a pipeline from upstream of the ODA into the historic Pole Canyon Creek channel downstream of the ODA, which has resulted in a dramatic improvement in water quality in Pole Canyon Creek.

2.4.6 Aquatic Biota

Perennial streams within and adjacent to the Site contain several species of fish (Table 2-6) and a wide variety of aquatic macroinvertebrates. Overall, the fishery appears to be in fair to good condition at most locations with adequate fish densities, good condition factors, few abnormalities, multiple life stages, and expected species diversity (NewFields 2009). While the uppermost portion of Smoky Creek does not have fish, lower reaches contain Yellowstone cutthroat trout (*Oncorhynchus clarkii ssp.*) (YCT), brook trout (*Salvelinus fontalis*), longnose dace (*Rhinichthys cataractae*), and sculpins (*Cottus spp.*) (BLM and USFS 2002). Seven species of fish were collected in the August 2000 sampling of Tygee Creek: cutthroat trout, brook trout, longnose dace, sculpin, redside shiner (*Richardsonius balteatus*), Utah chub (*Gila atraria*), and leatherside chub (*Gila copei*) (BLM and USFS 2002). During the 2010 RI sampling for Smoky Creek and Tygee Creek, YCT and Paiute sculpins (*Cottus beldingi*) were found. A more recent survey conducted by the University of Idaho in 2010 and 2011 found no leatherside chubs in Tygee Creek or Crow Creek, despite previously vouchered specimens from these streams (Keeley et al. 2012).

Mariah (1988) stated that mottled sculpin (*Cottus bairdi*) and speckled dace (*Rhinoichthys osculus*) were present in Tygee Creek and also expected in the Sage Creek drainage. Other fish that occur in the general area are brook trout, rainbow trout (*Onchorhyncus mykiss*), brown trout (*Salmo trutta*), and mottled sculpin, as well as other minnow species (Mariah 1988). Amphibian and reptile species known to occur in the Site include tiger salamander (*Ambystoma tigrinum*), boreal chorus frog (*Pseudacris maculata*), rubber boa (*Charina bottae*), and western terrestrial garter snake (*Thamnophis elegans*).

Sampling locations for aquatic biota are shown in Figure 2-8. In the Sage Creek and Crow Creek drainages, brown trout is the predominant salmonid species, followed by YCT, and mountain whitefish (*Prosopium williamsoni*). Mountain whitefish are found in lower Crow Creek and lower Sage Creek. Paiute sculpin has been almost exclusively found, with occasional mottled sculpins collected intermittently. Predominant cyprinid species include two species of dace, Longnose and Speckled. Redside shiner is more commonly found at the lower elevation Crow Creek locations downstream of Sage Creek. Catostomids are entirely comprised of Utah Sucker (*Catostomus ardens*) in the Crow Creek drainage. One leatherside chub was found in 2008 in an upper reach of Crow Creek. Dace species are typically found in the lower elevation Crow Creek areas whereas sculpin are predominant in the upper elevation reaches of Sage and Crow Creeks. Redside shiner and Utah Sucker are also found in the lower elevation reaches. Annual population monitoring from 2006 to 2012 has shown that mainstem Crow Creek locations both upstream and downstream of Sage Creek have the most diverse fish species assemblages, while tributary streams to Crow Creek (e.g., Deer Creek, Sage Creek) have fewer fish species (Table 2-6). Population estimates conducted annually from 2006 to 2012 indicate that Sage Creek has some of the highest trout biomass, even when compared to Crow Creek both upstream and downstream of Sage Creek (Figure 2-9).

2.4.7 Special Status Species

Literature and information obtained via correspondence with the USFWS, State of Idaho Department of Fish and Game (IDFG) (including Idaho Fish and Wildlife Information System [IFWIS]), USFS, and BLM, were evaluated to identify a list of threatened and endangered (T/E) and special-status species potentially present at the Site. Table 2-7 and Figure 2-10 provide a summary of T/E and special-status species identified for the region based on information obtained via correspondence (BLM 2009, IDFG 2009 and 2012, USFS 2010), as well as information obtained from literature sources (BLM 2010, Idaho Administrative Procedures Act [IDAPA] 2012, USFS 2011, USFWS 2013).

No USFWS threatened or endangered species are known to inhabit the Site or the immediate vicinity. As indicated in Figure 2-10, several species identified as at-risk by IDFG and IFWIS have been observed either within the boundaries of the Site (long-legged Myotis) or near the Site. While it is likely that one or more of these species may occasionally utilize the habitats available at the Site, none of these habitats are expected to be particularly attractive or of superior quality to these species compared with other available habitat in the vicinity of the Site.

No threatened or endangered aquatic species use streams that traverse the mine area. YCT, which is a designated sensitive species, have been found in several streams within and adjacent to the mine area. Northern leatherside chubs, a designated sensitive species, have not been found in streams within the mine area but have been reported in Tygee Creek, an adjacent stream north of the mine tailings impoundments.

2.5 Ecological Conceptual Site Model (ECSM)

The ECSM identifies the means by which ecological receptors may be exposed to Site contaminants and provides a basis for identifying data and risk analysis needs for the SSERA. The ECSM includes the following elements (USEPA 1997):

- Primary, secondary, and tertiary sources;
- Mechanisms of COPC releases from these source areas;
- Exposure pathways; and
- Receptor groups, exposure scenarios and assessment endpoints.

The ECSM for the Site is shown in Figure 2-11. The nature and extent of contamination by COPCs at the Site was characterized through the collection of a broad range of Site-specific physical and chemical information as detailed in the Final RI Report (Formation 2014). The RI evaluations showed and stated that, relative to all COPCs, selenium generally has the widest spatial distribution and greatest order-of-magnitude concentrations exceeding screening-level benchmarks for groundwater, surface water, sediment, and soil at the Site. Further, the RI showed that the extent of mining-related impacts is most easily identified by elevated selenium concentrations in Site media.

Overburden disposed in backfilled mine pits and in external ODAs is the source of selenium (and other COPCs) to the environment, as discussed in the Final RI Report (Formation 2014). Overburden is removed during active mining to access the underlying phosphate ore. The overburden contains different bedrock units (Meade Peak Member, Rex Chert Member, and Cherty Shale Member) that are chemically and mineralogically distinct from one another. The primary sources of COPCs within the overburden are the sulfides and organic matter present in the mudstone and middle waste shale from the Meade Peak Member. The release of COPCs from overburden materials to infiltrating/percolating water, with subsequent transport to Wells Formation groundwater and discharge to surface water via Hoopes Spring and South Fork Sage Creek springs is considered the primary mechanism for transport of selenium and other metals to the environment.

Backfilled pits and external ODAs have been reclaimed by various methods. The type of reclamation influences the relative importance of the source areas to release of selenium and other COPCs, with less protective covers, including direct revegetation, allowing larger contributions of COPCs to percolate to the underlying groundwater. For example, the ratio of infiltrating water to the mass of seleniferous overburden within an ODA is a factor in selenium rates of release and mass transport to groundwater flow systems. The Pole Canyon ODA is distinct from the other ODAs at the Site because of its cross-valley fill setting and the presence of an underlying shallow alluvial groundwater system associated with Pole Canyon Creek.

COPCs may be released from these areas via weathering and oxidation in soils and possible leaching into the groundwater. COPCs may then be further transported via wind dispersion/erosion, erosion via surface water runoff, surface water flow, or via groundwater. The potentially affected environment evaluated in the RI included terrestrial areas on the mine panels and ODAs, wetland and riparian areas along streams and seeps, and aquatic habitat in Sage Creek and downstream areas of Crow Creek.

The exposure pathways identified for the ECSM are consistent with those previously identified in the Area-Wide ERA (AWERA; TTEMI 2002) and in the SI Report (NewFields 2005), and reflect discussions between Simplot and the Agencies. The ecological receptors that are potentially exposed to the mine materials, or affected environmental media, include terrestrial and aquatic plants and animals on the Site and in downgradient areas.

The primary exposure pathway for plants is direct contact with contaminated media (rooting zone soil and water). For aquatic invertebrates and vertebrates, the primary pathways involve direct contact with water and sediment, and ingestion of contaminated food. The primary exposure pathway for vertebrate wildlife including birds, mammals, fish and reptiles, is ingestion of COPCs in food or water. This includes ingestion while feeding or drinking, and incidental ingestion from grooming and foraging behavior.

All RI soil and overburden samples were collected from the uppermost 6 inches of material (i.e., 0- to 6-inch depth interval). Overburden is run-of-mine (i.e., the material was placed as it was generated by mining with no sorting) and contains a mixture of different geologic materials with varying characteristics spatially. Selenium concentrations in overburden can be highly variable across short vertical and horizontal distances although, on a larger scale, no trends are observed over the surface and no trends are expected with depth.

The 0- to 6-inch depth interval was selected as the best balance for representing the depth interval to which wildlife would be directly exposed during feeding, preening or other behaviors; this depth also would represent concentrations in which a substantial proportion of grass and forb root mass would be located. Many vegetation species have roots that extend to greater depths, but will have a large root mass from which nutrients and water are absorbed in the 0- to 6-inch depth interval.

The most important potential effect of selenium on terrestrial wildlife is through ingestion of food items from the surface. Vegetation samples from potentially affected soils at the Site directly represent the potential exposure of herbivorous wildlife. The vegetation samples also represent the effect of subsurface soils from relevant depths on selenium concentrations in above-ground foliage to which wildlife would be exposed. The same is true for the invertebrate and small mammal samples collected from the Site. Therefore, samples of vegetation, small mammals, and invertebrates represent the potential impacts of surface and subsurface soils on exposure to wildlife receptors.

Other potentially-complete routes, but considered less likely to contribute to the exposure of ecological receptors, are inhalation of windblown particles and dust, and trans-dermal absorption.

The receptor groups to be evaluated in the SSERA are consistent with the ECSM in the AWERA, and are identified based on trophic level:

Terrestrial Receptors:

- 1° Consumers – Terrestrial Herbivorous Mammals; small receptors represented by Black-tailed jackrabbit (*Lepus californicus*) [Surrogate = Eastern Cottontail (*Sylvilagus floridanus*)] and ungulates represented by Mule Deer (*Odocoileus hemionus*).

- 1° Consumers – Terrestrial Herbivorous Birds; represented by Chipping Sparrow (*Spizella passerina*) [Surrogate = Northern Bobwhite *(Colinus virginianus*)].

- 2° Consumers – Terrestrial Omnivorous Birds; represented by American Robin (*Turdus migratorius*).

- 2° Consumers – Terrestrial Omnivorous Mammals; represented by Deer Mouse (*Peromyscus maniculatus*).

- 2° Consumers – Terrestrial Reptiles; represented by Western Garter Snake (*Thamnophis elegans*).

- 3° Consumers – Terrestrial Carnivorous Mammals; represented by Coyote (*Canis latrans*).

- 3° Consumers – Raptors; represented by Northern Harrier (*Circus cyaneus*).

Aquatic/Riparian Receptors:

- 1° Consumers and Producers – Non-fish aquatic life (including periphyton, zooplankton, aquatic plants, invertebrates, and amphibian tadpoles).

- 1° Consumers – Benthic fish; represented by Sculpin (*Cottus* sp.).

- 1° and 2° Consumers – Fish; represented by Cutthroat Trout (*Oncorhycus clarki*).

- 1° Consumers – Aquatic and Riparian Herbivorous Birds; represented by Song Sparrow (*Melospiza melodia*).

- 1° Consumers – Aquatic and Riparian Herbivorous Mammals; represented by Long-tailed Vole (*Microtus longicaudus*) [Surrogate = Meadow Vole (*Microtus pennsylvanicus*)].

- 2° Consumers – Non-Fish; Amphibians

- 2° Consumers – Aquatic and Riparian Omnivorous Birds; represented by Red-winged Blackbird (*Agelaius phoeniceus*).

- 2° Consumers – Aquatic and Riparian Piscivorous Birds; represented by Belted Kingfisher (*Megaceryle alcyon*).

- 2° Consumers – Aquatic and Riparian Omnivorous Mammals; represented by Raccoon (*Procyon lotor*).

- 2° Consumers – Aquatic and Riparian Carnivorous Mammals; represented by Mink (*Mustela vison*).

Potential risks to receptors listed above were quantitatively assessed in this SSERA. Although domestic livestock are included in the SSERA Work Plan (Formation 2011a), they are not considered “ecological receptors” and are evaluated independently for potential acute and chronic risk from selenium in the Site-Specific Livestock Ecological Risk Assessment (Formation 2015a).

2.6 RI Data Collection and RI COPCs

As part of the RI, samples of the following media were collected to estimate exposure in the screening-level assessment.

- Terrestrial, Riparian and Seep/Spring Areas
  - Soil;
  - Vegetation;
  - Terrestrial invertebrate tissue; and
  - Small mammal tissue.

- Streams/Drainages
  - Sediment;
  - Surface water;
  - Aquatic algae and macrophytes;

- o Benthic invertebrate tissue; and
- o Fish tissue.

Data were collected under the RI/FS Sampling and Analysis Plan (SAP) (Formation 2010), and Data Quality Objectives (DQOs), background information, and the SSERA Work Plan were presented in the RI/FS Work Plan (Formation 2011b). Data collection for the RI was designed to address data gaps identified in the RI/FS Work Plan (Formation 2011b). The data gaps were based on a review of the large body of existing data for characterization of environmental conditions at the Site and in consideration of the general objectives of the RI/FS. The data collected for the RI/FS have been used to refine the preliminary characterization of the nature and extent, and fate and transport, of RI COPCs in the environment and are being used to support risk assessments for human and ecological receptors. A detailed description of the investigations conducted under the RI, along with available data and data quality, is provided in Section 2 of the Final RI Report (Formation 2014).

The list of the media-specific RI COPCs upon which the RI was based is provided in Table 1-1. The SSERA is based on the evaluation of risk from the entire list of RI COPCs.

2.7 Deviations from the Baseline Problem Formulation

The following discussion identifies the key deviations, changes, or additions to this SSERA since the Baseline Problem Formulation was submitted and approved (Formation 2013).

Surface Water

Tier 2 surface water quality criteria from Michigan Department of Environmental Quality (Michigan DEQ) used as benchmarks for this SSERA were updated from those used in the BPF; the BPF cited values from 2009, whereas the SSERA used 2014 values. In addition, while the old 1988 aluminum criterion was used, a more up-to-date criterion value for aluminum from the literature was also included for later stages of risk characterization.

Sediment

The SSERA used the MacDonald et al. (2000) threshold effect concentration (TEC) values for screening even if a lower National Oceanic and Atmospheric Administration (NOAA) Screening Quick Reference Table (NOAA SQUIRT) value was available. The probable effect concentration (PEC) values (which were not included in the BPF) were also included as sediment effect thresholds. For ECOPCs that did not have thresholds from either of the above sources, additional literature reviews were conducted to develop thresholds from available toxicity literature.

Biological Tissue

No screening for fish tissues occurred in the BPF. A literature search and review of the potential fish tissue toxicity reference values (TRVs) presented in the BPF was conducted, and the fish tissue TRVs were augmented with additional data. Trout and other fish species tissue residue effect data were separated to allow for comparison of trout and sculpin fish tissue data to a range of TRVs.

No amphibian tissue data were collected as part of the RI process, and in subsequent correspondence with the USFS and other Agencies, a process was laid out to evaluate potential risks to amphibians through a review of the available literature and use of sculpin data as a surrogate. TRVs for amphibians were found and utilized for those comparisons. In addition, a review of the literature allowed for compilation and derivation of a tissue based threshold for benthic invertebrates to assess potential selenium effects.

2.8 Screening Level Ecological Risk Assessment (SLERA)

The list of RI COPCs included 22 chemicals and a SLERA was conducted as part of the BPF to identify the RI COPCs that clearly do not represent unacceptable risk, and for which no further analysis is necessary. The remaining chemicals were identified as the ECOPCs for which the more detailed risk assessment was conducted. The SLERA corresponds to Step 2 of the USEPA eight-step ERA process (USEPA 1997).

As defined in the SSERA Work Plan (Formation 2011a), the following overall steps were used to conduct the screen of RI COPCs and identify ECOPCs:

- Any RI COPC that was not positively detected in 5% of the samples in a particular media type (minimum of 20 samples per RI COPC) was excluded. Note that no RI COPCs were removed from consideration as ECOPCs using this step.

- For plants and terrestrial invertebrates, the maximum concentration for each RI COPC detected in Site surface soils was compared to toxicity-based screening levels for each receptor group. RI COPCs for which the maximum Site-wide concentration exceeds the respective screening level were retained for further analysis as an ECOPC. RI COPCs for which the maximum detected concentrations were lower than the applicable toxicity-based screening levels were not retained as ECOPCs.

- For surface waters and sediments, the maximum measured concentration for each RI COPC detected in Site surface water, sediment, and fish tissue was compared to toxicity-based screening levels for each receptor group. RI COPCs for which the maximum Site-wide concentration exceeds the respective screening level were retained for further analysis as an ECOPC. RI COPCs for which the maximum detected concentrations were lower than the applicable toxicity-based screening levels were not retained as ECOPCs.

- For mammalian and avian wildlife, screening-level exposure estimates were calculated for ingestion pathways using the maximum detected concentrations in relevant environmental media (surface water, soil, sediment, plants, invertebrates, small mammals, and fish). Only measured tissue concentrations were used to estimate screening level exposure. The exposures were compared to screening-level TRVs. RI COPCs for which the predicted screening-level exposure estimates exceeded the No Observed Adverse Effects Level (NOAEL) TRV were identified as ECOPCs and retained for assessment in the risk analysis.

Any RI COPCs that were detected in at least one sample and do not have screening levels available were retained as ECOPCs, and discussed as part of the uncertainty analysis. Unlike surface water and sediment, which have fairly well defined and widely accepted TRVs (e.g., state standards, federal criteria, and sediment quality guidelines [SQGs]) for most analytes, the data for fish tissue effects thresholds are not so clearly defined. In the BPF (Formation 2013), an initial table of potential fish tissues effects thresholds was presented but no tissue data were screened. In this section, those initial TRVs from the BPF were augmented with additional literature searches to provide a tissue effects threshold for as many of the analytes as possible in order to conduct the initial Site-wide screening.

For Site-wide screening of fish tissue data, the maximum tissue concentration for each analyte, regardless of species, was compared to the lowest no or low effect level used as the TRV. Two TRVs were defined, as described more fully in the subsequent sections; one for trout and one for other freshwater species for each analyte if the data were available. Similar to surface water and sediments, comparison of the maximum Site-wide concentration to the lowest tissue TRV was conducted. ECOPCs were identified as those parameters with concentrations exceeding the respective screening TRVs. Parameters with no identified TRVs were carried forward to risk characterization as parameters with uncertain risk potential.

The benchmarks used for screening of sediment, surface water, fish tissue, and surface soils are provided in Tables 2-8 through 2-11. The results of the screening process for those media are provided in Tables 2-12 through 2-15. All summary statistics for the Site-wide screening data are provided in Appendix A.

As specified in the SSERA Work Plan (Formation 2011a), ECOPC screening for mammalian and avian wildlife was conducted by estimating intake of RI COPCs from food, water, and soil; and then comparing the intake estimates to TRVs that represent NOAELs. This approach for the wildlife screen was used because measured concentrations of RI COPCs were available for all of the relevant media, reducing the uncertainty associated with screening on the basis of generic screening levels that do not reflect conditions at the Site.

The generic equation used to calculate intake is:

$$\mathrm{Dose}_{\mathrm{Total}} = (\mathrm{SUF}) \times \frac{\left[\left(\mathrm{C}_{\mathrm{media}} \text{ x } \mathrm{IR}_{\mathrm{media}}\right) + \left(\mathrm{C}_{\mathrm{prey}}\right)\left(\mathrm{IR}_{\mathrm{prey}}\right)\right]}{\mathrm{BW}}$$

Where:

$\mathrm{Dose}_{\mathrm{Total}}$= Daily dose resulting from ingestion of abiotic media and dietary items (milligrams chemical per kilogram body weight per day [mg chemical/kg BW/day]).

C_{media} = Maximum Concentration of chemical in abiotic media (milligrams per kilogram [mg/kg] or milligrams per liter [mg/L]) during incidental ingestion of that media.

C_{prey} = Maximum Concentration of chemical in prey or forage types (mg/kg).

IR = Ingestion Rate (the amount of prey items, surface water, sediment, and soil ingested per day) (kilograms per day [kg/day], kg/kg BW/day).

BW = Body Weight of receptor species (kg).

SUF = Site Use Factor to account for the amount of time that the organism spends using the Site.

The inputs to the exposure assessment model are presented on Table 2-16 and Table 2-17. These tables indicate the Site-specific dietary items and diet percentages used to calculate an estimated total intake for each receptor along with the remaining exposure factors used in the equation above. All exposure factors (e.g., tissue concentrations, ingestion rates) used in the exposure estimation are presented on a dry weight basis.

The exposure parameters, such as daily rates for intake of forage, prey, water, and incidental ingestion of media, used to develop the exposure assessment model are similar to those presented in the AWERA (but with updates to more current sources in some cases) and in the SSERA for the nearby Conda/Woodall Mountain Mine Site (Formation 2012a). These parameters are largely based on standard source documents (e.g., *Wildlife Exposure Factors Handbook* [USEPA 1993]). The exposure factors were selected during the AWERA process to represent a reasonable maximum exposure (RME). The SUF was assumed to be 1.0 for all receptors and the maximum Site-wide concentration of each RI COPC in each medium was used in the exposure model.

The total intake was then compared with a NOAEL TRV to assess whether the RI COPC was retained as an ECOPC. NOAEL TRVs are measures of effects that represent exposure levels at or below which no adverse effects are expected.

For the RI COPCs for which USEPA Ecological Soil Screening Levels (EcoSSLs) are available (USEPA 2005 and updates), the NOAEL TRV derived by USEPA and used in the EcoSSL calculation was used as the screening-level TRV for birds and mammals. For all other RI COPCs, NOAEL TRVs were selected as discussed in the SSERA Work Plan (Formation 2011a). The NOAEL TRVs are provided in Table 2-18.

The results of the wildlife SLERA exposure calculations are provided in Table 2-19 along with comparisons to the NOAEL TRVs.

2.9 Ecological Chemicals of Potential Concern

The results of the SLERA are provided as a list of ECOPCs for each media/receptor pair. Table 2-20 provides a summary of the ECOPCs selected for further assessment in the SSERA for aquatic receptors. Tables 2-21 and 2-22 summarize the ECOPCs selected for further assessment for terrestrial receptors.

All RI COPCs that were not identified as ECOPCs (or of uncertain risk) are assumed to be of *de minimis* risk to ecological receptors at the Site and are not discussed further in the SSERA.

For aquatic receptors, the following RI COPCs were identified as posing *de minimis* risk: antimony, cobalt, lead, mercury, and uranium.

For fish tissues, the following RI COPCs were identified as posing *de minimis* risk: antimony, arsenic, beryllium, boron, lead, mercury, molybdenum, silver, thallium, and uranium.

For terrestrial/wildlife receptors, only silver was identified as posing *de minimis* risk.

2.10 Assessment and Measurement Endpoints for the SSERA

As part of problem formulation, USEPA guidance (USEPA 1997) recommends identifying overall site management goals and assessment/measurement endpoints on which the analysis of risk should focus. Assessment endpoints are explicit descriptions of the ecological values to be protected as a result of management actions at a site. Measurement endpoints are specific data collected to address the assessment endpoints in an attempt to answer the risk questions as related to the risk management goals at a site.

Assessment and measurement endpoints associated with the potentially exposed receptor groups discussed are presented in Table 2-23.

Overall, significant adverse ecological effects are defined as toxicity from Site conditions that result in reductions in survivorship or reproductive capability, threatening populations or community function. For species that are afforded additional regulatory protection due to their rare or threatened status, significant adverse effects can occur even if individuals are affected. For other species with stable or healthy populations, the assessment focused on community-level or population-level effects where some individuals may suffer adverse effects, but the effects are not ecologically meaningful because the overall Site population is not significantly affected. Risk was assessed in terms of an ‘average reduction in survivorship and fecundity across a population of organisms’ for these species.

Risk to amphibians was assessed through evaluation of other non-fish aquatic life and risk to reptiles was assessed through evaluation of birds, since both non-fish aquatic life and birds are known to be sensitive to potential Site ECOPCs.

Measures of exposure are defined as those measures that describe the location and concentration of ECOPCs in abiotic and biotic media that are used to estimate exposure to ECOPCs for each receptor considered in the SSERA (USEPA 1998). Exposure was assessed in a tiered approach, as follows:

- Tier 1: Site-wide;
- Tier 2: By reclamation area (and Sage Valley) for terrestrial receptors and by drainage (listed below) for aquatic/riparian receptors; and
- Tier 3: By sampling location.

The drainages considered in the SSERA are:

- Smoky Creek;
- Roberts Creek/Tygee Creek;
- Pole Canyon Creek;
- North Fork Sage Creek and northern Sage Valley;
- Sage Creek;
- Hoopes Spring;
- South Fork Sage Creek;
- Lower Sage Valley; and
- Crow Creek.

The primary risk questions answered in the SSERA are:

1) Do ECOPC concentrations in upland and terrestrial habitats represent a significant source of risk capable of adversely affecting populations of common species and/or individuals of threatened or endangered species inhabiting the areas potentially affected by current or historical mining at the Site?

2) Do ECOPC concentrations in surface water, sediment, and/or aquatic receptor tissues represent a significant source of risk capable of adversely affecting the aquatic community and/or specific aquatic receptors inhabiting the water bodies and drainages downgradient of the Site?

3.0 AQUATIC RISK ANALYSIS AND CHARACTERIZATION

Risk analysis includes both exposure analysis and effects analysis. Exposure analysis is used to quantify the degree to which receptors are exposed to ECOPCs in each exposure domain. Effects analysis attempts to determine the relationship between exposure to ECOPCs and observed or potential effects to the assessment endpoints.

The estimation of exposure, effects, and the characterization of risk for aquatic receptors are discussed in detail in the following sections. The assessment and measurement endpoints associated with the potentially exposed receptor groups were discussed in Section 2.7 and form the basis for the exposure assessment and effects assessment to be completed under Step 6 of the USEPA ERA process (USEPA 1997).

Risk Characterization (Step 7), presented in Section 3.3, evaluates the results of the risk analysis and provides a comparison of the Site-specific exposures (Section 3.1) to toxicity benchmarks and TRVs (Section 3.2), and provides information on interpreting the results for the assessment endpoints (Figure 1-3).

3.1 Aquatic Exposure Analysis

Exposure results from contact between a receptor of concern and one or more COPCs in environmental media. In the exposure assessment portion of the SSERA, the magnitude of exposures is estimated for receptors. For exposure to occur, a potentially complete exposure pathway must be present including a release to an environmental medium, and a point where receptors could contact the affected medium. The ECSM (Figure 2-11) and the Site-wide screening level assessment indicated that complete exposure pathways exist, and that the maximum exposure concentrations for the following RI COPCs in surface waters and sediments are ECOPCs, deserving further evaluation:

- Surface water – aluminum, arsenic, cadmium, iron, nickel, selenium, and zinc.
- Sediment – arsenic, barium, beryllium, boron, cadmium, chromium, copper, iron, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, vanadium, and zinc.
- Fish tissue – aluminum, barium, cadmium, chromium, cobalt, copper, iron, manganese, nickel, selenium, vanadium, and zinc.

This exposure assessment provides a more detailed estimation of exposure including consideration of the distribution of contaminants, the average concentrations to which receptors may be exposed, and other factors such as bioavailability and frequency of contact that may affect the potential toxicity. The first step in characterizing exposure is estimating the exposure point concentrations (EPCs) for ECOPCs in the environmental media to which receptors may come into contact. Analytical results from soil, surface water, and sediment samples collected during the RI were used to estimate EPCs.

3.1.1 Derivation of Exposure Point Concentrations

Site-wide exposure point concentrations in abiotic and biotic exposure media were represented by the 95 percent upper confidence limit (95UCL) on the mean (i.e., $p<0.05$). Data that were reported as less than the detection limit were included in the EPC calculations. The statistical program ProUCL (USEPA 2013) was used to calculate the 95UCL from datasets that included non-detect values by allowing the program algorithms to estimate an upper confidence limit (UCL) through bootstrapping methods. The output from ProUCL recommends a 95UCL and that value was used unless it was greater than the maximum concentration. If the 95UCL was greater than the maximum concentration, then the maximum concentration was used as the EPC. Distribution testing and statistical calculations for estimating the concentration terms used to quantify exposure are provided, in detail, in the following subsections.

EPCs are derived for several different levels or tiers of the SSERA. Recall that the initial screening of RI COPCs compared maximum concentrations of analytes in environmental media to conservative thresholds or benchmarks. Tier 1 of the SSERA assesses ECOPCs on a Site-wide level by analyte using the 95UCLs as a more representative EPC. No ECOPCs are eliminated in the Tier 1 analysis. In the Tier 2 assessment, EPCs are derived on a drainage-wide scale to narrow the focus for each ECOPC within each of the specific drainage basins being evaluated. Finally, in the Tier 3 assessment EPCs are derived on a site-by-site basis in those drainages where the 95UCLs from the Tier 2 assessment indicate that the EPCs exceeded respective benchmarks. EPCs were not derived for amphibians as no amphibian data have been collected for the Site.

3.1.1.1 Tier 1 – Site-Wide EPCs

The Tier 1 Site-wide assessment using 95UCLs rather than maximum concentrations provides a better estimate of exposure conditions for aquatic populations and wider ranging species (e.g., migratory trout). It is not meant to be an exclusionary tier in the risk characterization process, thus no ECOPCs are eliminated during Tier 1 characterization.

<u>Surface Water</u>

Surface water EPCs derived across the Site for aquatic habitats are shown in Table 3-1. Based on the initial screening using maximum concentrations from aquatic habitats as described in

Section 2.7, aluminum, arsenic, cadmium, iron, nickel, selenium, and zinc were the analytes carried forward as ECOPCs for surface waters. Of particular note for the surface water EPC summary statistics, cadmium and zinc were infrequently detected (Table 3-1) given the 82 samples collected for each analyte.

Sediment

Similar to surface water, EPCs for sediment were derived for aquatic habitats across the entire Site (Table 3-2). Sediment ECOPCs identified from the initial screening level assessment included arsenic, barium, beryllium, boron, cadmium, chromium, copper, iron, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, vanadium, and zinc. Of these, no sediment quality benchmarks could be located for beryllium, boron, molybdenum, thallium and vanadium; therefore, these analytes were carried forward as ECOPCs due to uncertainties concerning the concentrations measured and no comparable thresholds to assess effects. Molybdenum was the only analyte with a higher frequency of non-detects than detections, when compared to the other analytes evaluated.

Fish Tissue

Fish tissue EPCs, for the available Site data, were organized by dividing the data into two groups for the two predominant genera found in streams in vicinity of the Site, which included trout species (Brown trout and YCT) and sculpins (Paiute and mottled sculpins) (Tables 3-3 and 3-4). Between the two genera, the higher EPCs were typically found in trout relative to sculpins. Exceptions to this included aluminum, barium, iron, manganese, mercury, and selenium where sculpins tended to have higher concentrations than trout.

3.1.1.2 Tier 2 – EPCs by Drainage

Table 3-5 shows the EPCs derived for surface water and Table 3-6 shows the EPCs derived for sediments for each of the ECOPCs carried forward. Table 3-7 presents the EPCs derived for trout tissues for each of the ECOPCs carried forward. No fish were found to be present from Roberts Creek, Pole Canyon Creek, or North Fork Sage Creek. Table 3-8 provides the EPCs derived for sculpin tissues for each of the ECOPCs carried forward.

3.1.1.3 Tier 3 – EPCs by Drainage and Locations within Drainages

The Tier 3 assessment is a two-part assessment, for the abiotic and biotic media sampled at the various sites for aquatic habitats. The first part presented here involves grouping of individual sites based on their location relative to potential inputs of ECOPCs or significant flow inputs. For example, sites on Crow Creek were grouped as upstream and downstream of Sage Creek, with the known input being Sage Creek. To the extent the data are available EPCs are derived for these site groupings, upstream and downstream of significant inputs, by drainage. A compilation of these sites is shown in Table 3-9 and the EPCs for the upstream and downstream groupings are summarized in Table 3-10 for surface water.

For sediments, the limited amount of data from sites does not allow for use of ProUCL to derive 95UCLs. Maximum concentrations were used for sediment comparisons in the risk characterization.

For fish tissues, because fewer sites were sampled than for surface water, a similar level of assessment for fish tissues to compare upstream and downstream tissue concentrations is not practical. The Tier 3 assessment of fish tissues is conducted on a site-by-site basis for the three primary species found: brown trout, YCT, and sculpins. EPCs for the ECOPCs for each of these species are provided in Tables 3-11, 3-12, and 3-13, respectively.

In the risk characterization section that follows, the second part of this Tier 3 assessment, which involves a site-by-site assessment of ECOPCs where risk was predicted for the grouped sites (e.g., Hazard Quotient [HQ]>1), is provided. Where adequate data were available, EPCs were derived using ProUCL to calculate 95UCL concentrations. Where adequate data were not available, maximum concentrations for a site were used. The Tier 3 assessment for each medium (surface water, sediments, and fish tissues) is developed in the risk characterization.

Selenium tissue data for sculpins from Table 3-13 are also used as a surrogate to assess selenium risks for amphibians, as described below in Section 3.2.1.3. Benthic invertebrate tissue data compiled from composite samples from each site are used to assess selenium risks to benthic invertebrates as described in Section 3.2.1.3 using the exposure data presented in Table 3-14.

3.2 Aquatic Effects Analysis

The effects assessment has two components. First, benchmark exposures, called TRVs, representing known levels of toxicity (or lack thereof), were developed based on scientific literature or other sources of toxicity data. The hierarchy for selecting TRVs for each medium is described more fully in the following sections. Exposure estimates were then compared to the TRVs to help determine whether exposures at the Site are potentially ecotoxic. The second component included direct measurement of biological endpoints to determine whether adverse effects predicted by the exposure and toxicity assessment were observed at the Site. Measuring the response of ecological endpoints may not be practicable or possible for each receptor group or ECOPC, in which case risk characterization is based primarily on comparison of estimated exposures to TRVs. The methods for estimation of exposure and selection of benchmark exposures are described in the following sections.

3.2.1 Toxicity Reference Values

This section presents the TRVs for surface water, aquatic sediment, and aquatic tissue. Numeric thresholds of toxicity were available for all surface water COPCs and most sediment COPCs, with the exception of beryllium, boron, molybdenum, thallium, and vanadium. Numeric values and sources of TRVs for sediment, surface water, and fish tissue are presented in Tables 2-8, 2-9, and 2-10, respectively.

3.2.1.1 Surface Water

Final TRVs for this SSERA were derived primarily from chronic Idaho State standards (IDAPA 58.01.02). Where standards were not available, USEPA Ambient Water Quality Criteria (AWQC) were used. In some cases, AWQC were also not available; for those cases, secondary chronic values (Tier II) derived by the Michigan DEQ (2014) were used. If a Tier II value was available and a USEPA approved State standard was available from another state, then the approved State standard was used. Secondary chronic values (Tier II) were derived similarly to AWQCs, but without the required response data from eight taxonomic families necessary to derive a Tier I value. Numeric surface water values used as TRVs are listed in Table 2-9. For those analytes with hardness-based criteria or standards, a hardness value of 127.2 mg/L calcium carbonate (CaCO3) was used, which represents the 5th percentile hardness value from all sites evaluated.

Chronic values were the first choice for use as TRVs because these values represent a threshold of acceptable effects levels over a long period of time. Continuous exposure of organisms over an extended period of time can affect survival, growth, reproduction, and physiological and biochemical internal processes. For most chemicals, chronic effects data represent threshold values that are sensitive. Chronic effects data may be derived based on effects thresholds for one or more life stages, including sensitive early life stages. In addition, sensitive endpoints such as growth and reproduction are often used. As a sensitive threshold for effects, use of chronic data for TRVs provides a conservative approach for estimating risk.

TRVs for arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc are chronic State of Idaho water quality standards. Of these, cadmium, copper, lead, nickel, silver, and zinc are hardness-based standards. Hardness-based standards account for hardness in the derivation of the value because hardness, among other factors, can affect the bioavailability and thus the toxicity of certain metals. As hardness increases, so does the derived hardness-based standard. For this SSERA characterization, and because multiple hardness values were collected over different flow conditions at each sample location, the 5th percentile hardness from all sites was initially used as a representative conservative screening hardness value from which to derive hardness-based TRVs.

Selenium is important at this Site. The State of Idaho surface water quality standard is 0.005 mg/l as is the case in most states of the U.S. A Draft National Selenium Criterion (USEPA

2015) was released in July 2015 as a public review draft to solicit public comments. The criterion is based on maternal bioaccumulation from dietary exposure, and the resulting developmental effect on fish embryos and early life stage fish. The new criterion is based on an effects threshold for egg/ovary tissue concentrations rather than selenium concentrations in water. Selenium accumulated via diet is stored in egg tissues, sometimes months in advance of oviposition, which can result in early life stage mortality and teratogenic effects that in turn can result in mortality.

The developmental effects at the sensitive early life stages of fish have been the focus of the Draft National Selenium Criterion (USEPA 2015) as well as numerous studies that have advanced the understanding of selenium effects in fish. Simplot's studies on maternal bioaccumulation and effects to young developing brown trout and YCT (Formation 2012) are included among these studies and are used in the 2015 Draft Criterion.

While this criterion focuses on egg/ovary tissues, it has important implications for how surface water criteria will be identified and used in the future. USEPA provided implementation procedures in the 2015 Draft National Selenium Criterion to allow for translation of egg/ovary effects thresholds to whole body tissue residues, and/or water column concentrations of dissolved selenium. Simplot's brown trout data utilized by USEPA in developing the Draft Criterion is particularly relevant to this Site because those data are the result of adult fish reproduction studies conducted for this Site as part of developing a Site-specific criterion for selenium for the Smoky Canyon Mine area streams.

For this SSERA, the selenium TRVs utilized include both the State of Idaho Water Quality Standard (0.005 mg/L) as a screening TRV, and in later stages, the egg/ovary effects threshold for brown trout derived and presented in USEPA's 2015 Draft Criterion document as well as Simplot-derived values from the brown trout study. Collectively, the tissue range for selenium presented and used for assessing risks in this SSERA provide the most scientifically defensible thresholds available for this Site. More detail on the brown trout studies and the development of the fish tissue TRV for selenium is provided in Section 3.2.1.3, below, as well as in Appendix D.

The benchmark for iron was taken from USEPA's (2014) National Water Quality Standards Table, which references USEPA's 1986 Goldbook, Quality Criteria for Water. Limited data were available then as they are now for this criterion; it is based on a single limited field study.

Aluminum TRVs include threshold values from two different sources. The first source is based on the chronic criterion from USEPA AWQCs (1988) for aluminum, which is valid for waters with a pH between 6.5 and 9. Aluminum solubility is highly affected below pH 6.5, with a subsequent increase in Al^{+3} as pH decreases; however, USEPA's 1988 criterion does not account for this. The current chronic AWQC is driven by brook trout and striped bass studies that were carried out on test waters with very low hardness (between 12 and 14 mg/L as CaCO3) at low pH, much lower than is typically observed in the western states.

Since the AWQC was developed in 1988, a number of additional acute and chronic aluminum toxicity studies have been published, many of which meet USEPA guidelines for AWQC development (Stephan et. al. 1985). Additionally, efforts by the Arid West Water Quality Research Project (AWWQRP, 2006) and a review by Parametrix (2009) have provided additional evidence that aluminum toxicity is not only pH-dependent but hardness-dependent as well. Using the new species toxicity data, recalculated hardness-based criteria for aluminum have been developed (Parametrix 2009). In 2012, USEPA Region 6 approved the use of the hardness-based equation in New Mexico. They stated, "Based on an extensive review of the supporting documentation, we are approving the application of the hardness-dependent equation for aluminum to those waters of the State at a pH of 6.5 to 9.0 because it will yield criteria that are protective of applicable uses in waters within that pH range" (USEPA 2012). These criteria are expressed using the following formulas:

$$\text{Acute Criterion (ug/L)} = e^{(1.3695[\ln(\text{hardness})]+1.8309)}$$

$$\text{Chronic Criterion (ug/L)} = e^{(1.3695[\ln(\text{hardness})]+0.9162)}$$

These revised criteria were developed using the standard process outlined in the federal guidance (e.g., Stephan et al. [1985], the same guidance used to derive the 1988 criteria). The process used meets standards outlined in Stephan et al. (1985), which included hardness as a factor affecting the toxicity of aluminum. As a result, these revised aluminum criteria are applicable to streams in the vicinity of the Smoky Canyon Mine despite being developed for Los Alamos National Laboratories in New Mexico as no assumptions were made specific to the arid west in the derivation of this criterion. The revised aluminum criteria were not developed with specificity only to New Mexico waters or species.

Use of the chronic National AWQC to evaluate potential aluminum risk to the aquatic community is overly conservative, especially in waters with higher hardness and pH. This AWQC for aluminum is based on minimal species data that do not consider hardness and were conducted at the low end of the pH range which the 1988 criteria are reported to represent. The information presented above recognizes these limitations and adds additional toxicity data to a recalculation of the aluminum criteria values.

Michigan DEQ (2014) derived secondary chronic criteria for barium, beryllium, boron, cobalt, molybdenum, thallium, and vanadium. These secondary ("Tier II") values tend to be highly conservative.[1] Because fewer toxicity tests than the number of tests used to derive National criteria are used in the derivation of a Tier II value, an additional safety factor is used in the derivation process. Higher numbers of test results allow for the use of a lower safety factor, while lower numbers of test results require using a higher safety factor.

[1] Michigan DEQ frequently revisits and revises its Tier II Rule 57 values based on the availability of new toxicity data for different parameters. The process is conducted under the Great Lakes Water Quality Initiative. Suter and Tsao (1996) values were developed using the same process as Michigan's Tier II values, but have not been updated since 1996.

Hardness-based TRVs for manganese and uranium were obtained from Colorado Department of Public Health and Environment standards for the Arkansas River (CDPHE 2007). These values are used as state water quality standards and have been approved for use by USEPA Region 8.

3.2.1.2 Sediment

Numerous chemical-specific benchmarks for sediment are available in the literature. For most of the ECOPCs identified in this SSERA, toxicity-based screening benchmarks were found with the exception of beryllium, boron, molybdenum, thallium, and vanadium.

Screening and risk characterization through each tier was conducted primarily using sediment quality guidelines (SQGs) presented in MacDonald et al. (2000) which are widely accepted and based on empirical data from a wide range of testing conditions. SQGs presented in MacDonald et al. (2000) define TECs, which are concentrations below which no effects are expected, and PECs, which are concentrations above which statistically detectable toxic effects are expected. TECs and PECs are defined for several parameters, including arsenic, cadmium, chromium, copper, lead, nickel, silver, and zinc based on sediment dry weights.

For parameters where no SQGs were available, sediment thresholds from NOAA SQUIRTS (Buchman 2009) were used. If sediment thresholds were not available from either of the above two sources, literature searches were conducted after completion of the BPF (Formation 2013) to compile chemical-specific benchmarks. Through this process, sediment threshold benchmarks were identified for all ECOPCs except beryllium and vanadium (Table 2-8). Generally, sediment concentrations less than or equivalent to TEC values are not considered to pose a risk. Sediment concentrations that fall between the TEC and PEC values may pose an uncertain level of risk, while sediment concentrations greater than or equivalent to the PEC values typically pose a risk to benthic invertebrates. The uncertainties of potential risks, particularly for levels between the TEC and PEC values will be further discussed in the uncertainty section. Likewise, the uncertainty section will include discussion of those ECOPCs without either defined TECs or PECs, or similar type of values, or those that have no sediment benchmarks.

There are several different methods for deriving benchmarks and there is variability in the endpoints and responses used. Each approach has certain advantages and limitations that influence their application in the sediment quality assessment process (MacDonald et al. 2000). The majority of the available benchmarks for sediments have been developed based on invertebrate responses to chemicals in sediments.

The primary sources of literature reviewed in developing benchmarks included the following documents:

- MacDonald et.al. (2000) – Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems.
- Buchman (2009) – NOAA SQUIRTS guidelines.
- Ingersoll et al. (1996) – Calculation and evaluation of sediment effects concentrations for the amphipod *Hyallela azteca* and the midge *Chironomus riparius.*
- MacDonald et al. (2003) – Development and Evaluation of Numerical Sediment Quality Assessment Guidelines for Florida Inland Waters: Technical Report.

Methods for determining sediment benchmarks vary widely in the species used, exposure regimes, endpoints, and interpretation of data. Most sediment benchmarks are derived based on responses of invertebrate taxa such as amphipods, midges, mayflies, oligochaetes, daphnids, various bivalves, and bacteria. This is primarily due to the fact that such taxa are in intimate contact with sediments and control of exposures is both precise and accurate, making interpretation of results more straightforward.

Endpoints used in testing range from survival, growth, deformities, and reproduction to more subtle effects such as changes in biochemical biomarkers. Testing includes both field and laboratory exposures of organisms to individual chemicals and mixtures of chemicals.

Aluminum, manganese, iron, and silver TRVs were compiled from the NOAA SQUIRTS guidelines (Buchman 2009). An additional manganese and silver sediment toxicity value, apparent effects threshold (AET) was also found in the Washington guidelines (Cubbage et al. 1997). For silver, the Washington guidelines confirmed the value present in Buchman (2009), whereas for manganese an additional AET was defined.

The barium TRV was from USEPA (1977) which compiled values for evaluating Great Lakes Harbor sediments. The cobalt TRVs were derived from two sources including Ontario's open water disposal guideline (Persaud et al. 1993) and the USEPA (2003) derived lowest cleanup goal at the Blackbird Mine where cobalt was one of the primary COPCs.

The selenium TRV from Lemly (2002) has been widely cited as a benthic effect threshold. However, as noted in Lemly (2002), the 2 mg/kg value is cited as a potential threshold for bioaccumulative effects to higher trophic levels not as an effect threshold for toxic effects to benthic invertebrates. While this is a conservative benchmark for screening sediment concentrations of selenium, it does not provide an adequate benchmark for assessing the potential risks of selenium in sediments to aquatic invertebrates.

Van Derveer and Canton (2007) suggested, based on their analysis of selenium concentrations from numerous western streams and rivers, that a conservative threshold for potential effects of selenium to fish and wildlife is 4.0 mg/kg. This value represents the 10^{th} percentile of observed effects from the data evaluated. They found that organic carbon binds selenium allowing accumulation in stream sediments. In low organic carbon systems such as western streams, selenium accumulation in sediments is reduced as compared to those sediments evaluated by Lemley (2002).

While both are conservative estimates for potential effects, relative to this Site, the Lemly (2002) value is more conservative than the Van Derveer and Canton (2007) value. Lemly (2002) arrived at a value to be protective of higher-order consumers largely based on work in reservoirs in the eastern U.S. where selenite dominated these systems. Selenite bioaccumulates much more readily than selenate; selenate is more common in flowing streams of the western U.S. as found at this Site. Finally, for both values, the fundamental pathway for exposure is dietary uptake: water to algae, algae to benthos, benthos to fish, fish to fish, etc. Thus, the primary bioaccumulation mechanisms important for and appropriate for remedy decisions are the foodweb interactions and accumulation in algae. Sediment plays a role in this process, albeit small.

The Lemly (2002) value is used as an initial conservative sediment screening value for selenium. In Tier 1 and 2 risk characterization, Lemly (2002) and Van Derveer and Canton (2007) values are used to further refine potential sediment risks. Both values are primarily intended to evaluate potential developmental effects of selenium bioaccumulation in fish, which is widely recognized as the most sensitive aquatic environmental endpoint (USEPA 2015). However, bioaccumulation of selenium into the higher trophic levels is much less a function of selenium concentrations in sediments and much more a function of selenium concentrations in the primary producers (e.g., algae and periphyton) which is the largest bioconcentration step for selenium in aquatic foodwebs. The screening levels are not necessarily indicative of risk to invertebrates, which tend to be more tolerant. Therefore, tissue TRVs are also used to evaluate risk for invertebrates. Tier 3 risk characterization will examine those locations where selenium concentrations in sediments exceed the sediment risk thresholds, and will compare invertebrate tissue concentrations for selenium to invertebrate tissue TRVs to assess if the sediment thresholds used herein provide for an accurate characterization of risk.

3.2.1.3 Aquatic Tissue

Fish Tissue

The TRVs for fish tissue were derived from two primary compilations. Jarvinen and Ankley (1999) compiled fish tissue effects thresholds for a number of parameters across a wide range of species depending upon the availability of data from the literature. Similarly, an electronic database maintained by the U.S. Army Corps of Engineers, called the Environmental Residue Effects Database (ERED), contains many of the same studies cited in Jarvinen and Ankley (1999) as well as some more recent data from the literature. Because both of these sources represent compilations of other published and some unpublished works, when possible the original study was obtained.

The hierarchy for selecting fish tissue TRVs is relatively straightforward. Information for each medium was culled to focus on freshwater fish species. Whole body tissue data was preferred over organ tissue data, and aqueous and/or dietary exposures were selected over injection studies. Because bioaccumulative effects are manifested over a longer exposure period, studies with longer exposure durations were preferentially selected over short-term chronic or acute studies. The information was further culled to low or no effect thresholds based on survival, growth, or deformities; biochemical effects were only considered if no other data were available. Fish tissue TRVs are shown in Table 2-10.

The most important fish tissue TRV is for selenium because selenium is the ECOPC that is the likely risk driver for this Site. As noted previously, USEPA has released a Draft National Selenium Criterion (USEPA 2015) that focuses on effects to young developing fish; the tissue of choice for measuring this endpoint is egg/ovary tissue. The effects concentration for egg/ovary tissue cited in USEPA (2015) for brown trout is 18.09 milligrams per kilogram dry weight (mg/kg dw) based on larval survival. Because Simplot conducted the brown trout adult reproduction studies using adults from various locations associated with this Site (Formation 2012c), these data are particularly relevant to this SSERA. An additional site-specific threshold was developed from these data which is based on a combined endpoint of larval survival and proportion of normal to deformed fish (e.g., surviving and normal fish) which equals 20.5 mg/kg dw selenium in eggs. See Appendix D for a more thorough discussion of how these tissue thresholds were derived, how the peer review was conducted, and the applicability to this Site.

Most of the data on selenium in fish tissues from the RI data collection from this Site are whole body tissue concentrations. Therefore, relationships between egg/ovary and whole-body samples derived for the brown trout used in Simplot's study are used to convert egg/ovary tissue concentrations to whole body tissue equivalent values. The whole body equivalent for brown trout using the range of egg effect tissue concentrations noted above (e.g., 18.09 and 20.5 mg/kd dw egg selenium) are 12.48 and 14.14 mg/kg dw whole body selenium.

The lower whole body value was used to screen selenium in fish tissues to identify sampling locations and stream reaches for more detailed analysis. The upper whole body value was used in the risk characterization to provide a more complete evaluation of risks of selenium in fish tissues for the Site.

Benthic Tissue

Benthic invertebrate tissue residues are not typically used to quantify risks. However, because selenium is the primary ECOPC at this Site and because tissue data are available from most locations evaluated, the literature was reviewed to determine if an appropriate benthic invertebrate tissue TRV could be located.

Conley et al. (2009) conducted a dietary feeding study on uptake of selenium in mayflies. Measureable effects on fecundity were found at dietary concentrations of selenium less than 11 mg/kg. The diet was comprised of algae which concentrate selenium several times the abiotic concentrations and also convert selenium into methylated forms which are much more bioavailable. Conley et al. (2009) demonstrate that, like fish, benthic invertebrate exposure to and effects from selenium are based on the dietary intake. Using the bioaccumulation factor of 2.2 provided by Conley et al. (2009), the 11 mg/kg dietary value corresponds to an adult mayfly tissue selenium concentration equal to 24.2 µg/g dw. In subsequent work, Conley et al. (2011) found that bioaccumulation and influence of selenium on mayfly performance may be tied to resource availability and quantity. Conley et al. (2013) reported a bioaccumulation or trophic transfer factor of 2.1 and defined secondary reproductive effects at a dietary concentration of 12.8 mg/kg dw, thus supporting their earlier work that effects occur at dietary concentrations greater than 11 mg/kg dw. Again, using the bioaccumulation factor and applying that to the dietary concentration of 12.8 mg/kg dw, a whole body tissue threshold of 26.9 mg/kg dw was derived.

Because selenium concentration data for benthic tissues were available from most locations, potential risks can be characterized by comparing empirical benthic tissue data from the Site to the Conley et al. (2009 and 2013) no and low effect dietary thresholds derived above for whole body tissues as part of the Tier 3 risk characterization. This characterization will be compared to those locations where sediment thresholds for selenium were exceeded to provide for a more accurate assessment of risks to benthic invertebrates.

Amphibian Tissue

Data available for developing TRVs and assessing risk to amphibians is very limited when compared to invertebrates and fish. Risk to amphibians was assessed using two of the lines of evidence (LOEs) used for fish: water quality (i.e., RI COPC concentrations in surface water) and potential tissue residues in adult and larval amphibians. The overall approach to assess amphibian risk to ECOPCs included conducting a literature search and review to determine whether information is available that indicates amphibians are either more sensitive to ECOPCs than fish (i.e., experience toxicity at lower exposure levels), or are likely to accumulate higher ECOPC concentrations. For potential amphibian tissue residues, since no amphibians were collected during the RI, it was agreed in discussions with the agency reviewers of this SSERA to use sculpin tissue data as a surrogate if a suitable amphibian effect threshold could be found. Sculpin tissue data from the Site were agreed to be a good surrogate for aquatic-feeding amphibians due to a similar range of dietary components, a relatively small home (feeding) range, and the availability of data from sculpin for most aquatic sampling locations.

Recent reviews of scientific literature suggest that amphibians are less sensitive to the effects of selenium than are fish (Kerby et al. 2010, Weltje et al. 2012). Kerby et al. (2010) evaluated a large number of exposure and toxicity tests including invertebrates, fish, and amphibians and found that amphibians may be less sensitive than other aquatic biota.

Weltje et al. (2012) conducted a comparative analysis of acute and chronic sensitivity of fish and amphibians for approximately 50 chemicals, including some metals, but mostly organic chemicals. Of the ECOPCs, only cadmium, copper, and zinc were evaluated. The study compared chronic no observed effects concentrations (NOECs) reported in the literature and/or regulations of various agencies. They found that amphibian NOECs were generally higher than sensitive fish species. The authors concluded that NOECs and water quality criteria generated for fish species will be generally protective of amphibians. They also concluded that additional amphibian testing may not be necessary for chemical risk assessment.

An overall conclusion from Kerby et al. (2009) and Weltje et al. (2012) is that amphibians are generally less sensitive than fish or other aquatic organisms to a broad range of environmental contaminants in water. However, neither of these reviews included dietary pathways that are important for exposure of aquatic vertebrates to selenium. Hopkins et al. (2006) examined developmental effects of selenium accumulation in maternal adults and transfer to developing embryos in eastern narrow-mouthed toads (*Gastrophryne carolinensis*). Female adult toads would have obtained most of the selenium body burden through dietary pathways. Similar to fish, selenium accumulated by the maternal parent is transferred to eggs and can affect developing young. The highest selenium accumulation in eggs (up to 80 to 100 mg/kg dw) was substantially higher than for trout eggs. Egg viability was higher, and deformities were lower (96 hour) than for reference eggs for all but one endpoint (craniofacial). These data suggest that *G. carolinensis* embryo development is less sensitive than brown trout to selenium in eggs. However, small samples sizes at the higher concentrations may have affected the ability to detect statistical differences.

Unrine et al. (2007) evaluated metal concentrations in mollusks, insect larvae, bullfrog tadpoles, and fish collected from a coal-ash affected swamp area of the U.S. Department of Energy Savannah River Site in South Carolina. Bullfrog tadpoles (*Rana catesbeiana*) accumulated between 1 and 4 times higher concentrations of several metals than other invertebrates and fish. For selenium, concentrations (whole body) in tadpoles were marginally higher (~1.5x) than concentrations in aquatic insect larvae (dragonfly genera *Tramea* and *Erythemis*), smallmouth bass (*Micropterus salmonoides)* and spotted sunfish (*Lepomis punctatus*)*.* The swamp collection site from which these data were collected is a lentic system, and the pattern of relative concentrations among these groups may not be comparable to the lotic systems at the Site. However, the similar concentrations among the tadpoles and other aquatic biota suggest that anuran amphibians will not bioaccumulate substantially higher selenium concentrations than the brown trout at the Site.

The available information suggests that surface water concentrations that are protective of fish are also protective of amphibians. Data on bioaccumulation and developmental toxicity for metals suggest that at least anuran amphibians do not accumulate substantially higher concentrations of metals than fish, and that tissue-based TRVs for fish are protective of the amphibians. Tissue concentration data and corresponding effects information for amphibians are extremely limited. While few data are available to set TRVs for amphibians based on tissue concentrations, further review of the literature indicates that amphibians can bioaccumulate selenium, selenium is maternally transferred to eggs, and effects are manifested in developing young. Interpretation of the Hopkins et al. (2006) study reveals an estimated NOAEL threshold value of about 20 mg/kg dw[2] can be derived. In a more recent study, Masse et al. (2014 unpublished, but cited in USEPA [2015]) derived an EC_{10} for the *Xenopus laevis*, a toad that is a standard test species in the Frog Embryo Teratogenesis Assay Xenopus (FETAX) toxicity assessment procedures. The study identified an EC_{10} value of 24.8 mg/kg dw in eggs and reported a 1:1 ratio of selenium in eggs and whole body, thus whole body concentrations at the effect threshold would also be 24.8 mg/kg dw.

These tissue effect thresholds will be used to assess potential selenium risks to amphibians using sculpin tissue data as a surrogate for amphibian data since no specific data on COPC concentrations in amphibians were collected for the Site.

3.2.2 Field Effects Data and Additional Lines of Evidence (LOE)

While not collected as part of the RI, data have been collected to assess the condition of the aquatic communities within, upstream, and downstream of the Site. Data on the structure of the fish and benthic macroinvertebrate communities, and physical habitat characteristics were collected during low- and high-flow conditions at sites upstream and downstream of the Site

[2] When all developmental criteria were considered collectively, offspring from the contaminated site experienced 19% lower viability, although egg Se and egg viability were not statistically related (Hopkins et al. 2006). While a true effects threshold related to amphibian body burdens was not derived in this study, there was a demarcation of effects relative to controls at the contaminated sites. The mean value of 42.4 mg/kg dw in whole body tissues has a large degree of uncertainty associated with it based on the Standard error presented. The mean value (n=10) for the contaminated sites was based on data spanning a wide range of body burdens, and Hopkins et al (2006) state that their statistical power for detecting functional relationships between concentrations and effects was probably limited within the range of concentrations where effects should be predominant (e.g., egg selenium concentrations > 20 mg/kg dw).

influence. These data, particularly the fish population estimates and benthic invertebrate community metrics, are used as additional LOEs to assess whether the aquatic community is being adversely affected under current conditions, the extent to which the community may be benefited or adversely affected under various remediation options, and to support application for permits that may be necessary for implementation of remedial actions.

Additional LOEs are also considered in conjunction with the hazard quotient (HQ) results as appropriate for all assessment endpoints. The SSERA Work Plan (Formation 2011a) presented a list of potential LOEs that may be used in the risk characterization. One particularly important LOE is the fish population data collected at numerous sites and streams since 2006. Trends that may occur in these fish populations, age frequency distributions, or shifts in species abundance or types in potential response to changing selenium levels in the environment will be critical in aligning the effects data from laboratory studies to actual population level effects in the field.

In addition, the Presser and Luoma (2010) selenium bioaccumulation model is used to develop estimates of aqueous selenium concentrations based on the food chain Site-specific trophic transfer factors (TTFs) and the Site-specific effects threshold for fish. Details for deriving an aqueous selenium concentration from the trophic transfer data and effects threshold are shown in Appendix D.

As part of the risk characterization, the primary sources of uncertainty are discussed and the potential effect of each uncertainty on the risk characterization is presented in the SSERA. Potential impacts of habitat change or loss, including co-existing chemical, physical, and biological stressors, are discussed as other potential LOEs.

Background considerations may also be incorporated into the assessment and investigation of sites, as acknowledged in existing USEPA guidance including *Role of Background in the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) Cleanup Program* (USEPA 2002a) and *Guidance for Comparing Background and Chemical Concentrations in Soil for CERCLA Sites* (USEPA 2002b). The guidance indicates that, although background should not be considered in identification of ECOPCs, it should be considered in risk characterization. As discussed in the SSERA Work Plan (Formation 2011b), no additional data collection to determine background concentrations of ECOPCs for this SSERA is planned. However, specific sampling and analysis activities needed to characterize background may be developed if the potential need and appropriate focus and scope of such sampling is identified based on preliminary risk characterization in the following section.

3.3 Aquatic Risk Characterization

The risk characterization phase of the ERA process is the point at which information on nature and extent of contamination, the exposure assessment, and the effects assessment are integrated to characterize risks to assessment endpoints (USEPA 1997, 1998). In this section, estimates of exposure are compared to TRVs to estimate the potential for adverse effects for each of the ECOPCs. In addition, direct measures of the biological communities at the Site are examined to determine whether adverse effects are observable and to assess correlation of effects with trends in chemical concentrations and seasonal variation. These two LOEs are then integrated to help determine the potential for adverse effects at the Site, the likelihood that the effects result from Site-specific releases or conditions, and the primary conditions contributing to effects and/or risk.

The risk characterization presented in this SSERA reflects exposure conditions present during 2010 and 2011 when samples were collected for the RI. These data are augmented with environmental data collected prior to and after this time frame in order to examine trends in ECOPCs, both spatially and temporally, and how those trends relate to field data for fish populations and benthic invertebrate communities.

Estimating risk based on exposure is conducted by comparing EPCs (or doses) derived in the analysis step with the media and or receptor-specific TRVs. Results are expressed as HQs (USEPA 1997):

$$HQ = \text{Exposure Point Concentration} \div TRV$$

If the HQ is equal to or less than 1 (indicating the exposure concentration or dose is less than the TRV), the occurrence of adverse effects is unlikely. If the HQ is greater than 1 (indicating the exposure is equal to or greater than the TRV), there is a potential for adverse effects to occur (USEPA 1997). However, there is no clear consensus from either USEPA guidance or the scientific literature concerning the significance of the level of departure from 1.

If an HQ cannot be calculated because insufficient data are available to establish a toxicity threshold, ECOPCs are carried through the risk characterization as ECOPCs of uncertain risk. These ECOPCs are qualitatively discussed in the Uncertainty Analysis (Section 5).

3.3.1 Tier 1 and 2 Baseline Assessment

Tier 1 exposure analysis of the SSERA is based on comparison of Site-wide EPCs to conservative TRVs. For most media in the aquatic assessment of this SSERA, the Tier 1 assessment reflects what was observed in the screening-level assessment where maximum Site-wide concentrations were compared to conservative TRVs.

3.3.1.1 Surface Water

Comparisons of Site-wide river habitat EPCs to TRVs for Tier 1 are shown in Table 3-15. The ECOPCs identified are similar to those identified in the screening-level assessment: aluminum, cadmium, nickel, selenium, and zinc. Arsenic and iron, which were identified as ECOPCs from the initial screening were found, based on their individual EPCs, to be lower than the screening TRVs. These ECOPCs together with the previously identified ECOPCs will be evaluated further in the Tier 2 analysis.

Tier 2 analysis involved deriving EPCs for sample location data for each ECOPC, by the respective drainages within which they occur (Table 3-16). Ten different drainages were identified. From this analysis the following observations can be made:

- EPCs for each ECOPC exceed their respective criteria in at least one drainage.

- Aluminum is an ECOPC that is distributed Site-wide. Only Hoopes Spring surface waters had concentrations of aluminum that were lower than the chronic criterion (0.087 mg/L). $HQ_{chronic}$ values among all drainages, excluding Hoopes Spring, ranged from 1 to 24. HQ_{acute} values greater than 1 were derived for Tygee Creek, North Fork Sage Creek, Sage Creek, and South Fork Sage Creek.

- Concentrations of dissolved arsenic, cadmium, nickel, and zinc were identified as ECOPCs for Pole Canyon Creek. $HQ_{chronic}$ values for each of these ECOPCs in Pole Canyon Creek were the only HQs greater than 1 for these parameters across any of the drainages. Arsenic and nickel HQ_{acute} values for Pole Canyon Creek were less than 1, while cadmium and zinc HQ_{acute} values were both greater than 1 in Pole Canyon Creek.

- The HQ for iron was greater than 1 in North Fork Sage Creek, while all other HQs for iron were 1 or less. No acute surface water TRV is available for iron.

- Selenium EPCs exceeded the current Idaho chronic standard in the Pole Canyon Creek, North Fork Sage Creek, Hoopes Spring, South Fork Sage Creek, Lower Sage Creek, and Crow Creek drainages. $HQ_{chronic}$ values ranged from 2 in North Fork Sage Creek to 680 in Pole Canyon Creek. In Pole Canyon Creek and Hoopes Spring, the acute HQs were greater than 1, ranging from 170 to 2, respectively. Selenium in each of these drainages is evaluated further in the Tier 3 assessment.

Based on the results of the Tier 2 risk characterization of surface water ECOPCs by drainage, the following ECOPCs and drainage combinations are carried forward to the Tier 3 risk characterization:

- Aluminum - All drainages except Hoopes Spring;
- Arsenic – Pole Canyon Creek;
- Cadmium – Pole Canyon Creek;
- Iron – North Fork Sage Creek;
- Nickel – Pole Canyon Creek;
- Selenium – Pole Canyon Creek, North Fork Sage Creek, Hoopes Spring, South Fork Sage Creek, lower Sage valley, and Crow Creek; and
- Zinc – Pole Canyon Creek.

3.3.1.2 Sediment

Sediment ECOPCs carried forward from the initial screening analysis included: arsenic, barium, beryllium, boron, cadmium, chromium, copper, iron, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. Of these, beryllium, boron, molybdenum, thallium, and vanadium had no screening-level TRVs. For sediments, two levels of TRVs are utilized, the TEC or similar values and the PEC or similar values. For the purpose of presentation in this SSERA, TECs or similar values are referred to as TRV_{low} values while PECs or similar values are referred to as TRV_{high} values. These do not necessarily imply low or high level of effects, rather lower and upper level threshold values that are used to clarify which TRV is being used when developing the HQs.

In the Tier 1 assessment using 95UCLs as the EPCs for each ECOPC, all ECOPCs identified from the initial screening are also identified in the Tier 1 assessment as exceeding the respective TRVs, with the exception of iron and mercury (Table 3-17). All of the ECOPCs were carried into the Tier 2 assessment to evaluate where ECOPCs concentrations pose a risk to receptors.

The Tier 2 assessment focuses on those ECOPCs with available TRVs. As discussed in Section 3.2.1.2, additional toxicity-based TRVs were identified for sediments from the literature and are used as part of the more focused Tier 2 assessment. No TRVs were identified for beryllium or vanadium.

Results of the Tier 2 analysis (Table 3-18) are as follows:

- EPCs for arsenic in sediments only exceeded the TRV_{low} in the Pole Canyon Creek drainage (HQ = 2) while none of the sediment EPCs from other drainages exceeded the TRV_{high}. All HQs for arsenic in sediments using the TRV_{high} were less than 1. Some limited sediment toxicity in Pole Canyon Creek due to arsenic may exist; arsenic for Pole Canyon Creek is carried forward for further risk characterization.

- Barium EPCs in sediments from all drainages exceeded the TRV_{low}. Sediment EPCs from Roberts Creek, Hoopes Spring, South Fork Sage Creek, and Crow Creek were equal to the TRV_{high} (HQs = 1) while EPCs in sediments from all the remaining drainages had HQs greater than 1 relative to the TRV_{high}. Barium is a Site-wide ECOPC, but is not found in the primary source area (e.g., Hoopes Spring). Barium in sediments for those drainages identified as uncertain is carried forward for further risk characterization.

- A toxicity-based TRV_{high} for boron in sediments was identified for the Tier 2 assessment. All sediment EPCs from all drainages were less than the TRV_{high} resulting in HQs less than 1.

- Cadmium EPCs in sediments from all drainages exceeded the TRV_{low}. Only Pole Canyon Creek and Sage Creek had cadmium EPCs that exceeded the TRV_{high} (HQs = 9 and 2, respectively). Cadmium in sediments for those drainages identified as uncertain or where HQs exceeded 1 are carried forward for further risk characterization.

- Sediment EPCs for chromium resulted in HQs greater than 1 relative to the TRV_{low} in Pole Canyon Creek and Sage Creek; however, only in Pole Canyon Creek did the chromium EPC exceed the TRV_{high} (HQ = 3). All other drainage EPCs were less than an HQ of 1 relative to the TRV_{high}. Chromium in sediments for those drainages identified as uncertain or where HQs exceeded 1 is carried forward for further risk characterization.

- Copper EPCs only exceeded the TRV_{low} in the Pole Canyon Creek drainage (HQ = 2). No sediment EPCs exceeded the copper TRV_{high} with all HQs being less than 1. Copper in Pole Canyon Creek is carried forward as an ECOPC for further risk characterization.

- Iron EPCs in sediments resulted in HQs of 1 or less relative to the TRV_{low}. No iron EPCs exceeded the TRV_{high}; all HQs were less than 1. Iron is not carried forward for further risk characterization in sediments.

- Manganese EPCs in sediments from all drainages except Roberts Creek, Hoopes Spring, and Crow Creek resulted in HQs greater than 1 relative to the TRV_{low}. Compared to the TRV_{high}, only the North Fork Sage Creek and Lower Sage Creek manganese EPCs resulted in HQs greater than 1 (HQs = 5 and 2, respectively). Manganese in sediments from all drainages except Roberts Creek, Hoopes Spring, and Crow Creek is carried forward for further risk characterization.

- Mercury EPCs did not result in HQs greater than 1 relative to the TRV_{low}. Mercury is not carried forward for further risk characterization.

- All molybdenum HQs were less than 1 relative to the TRV_{high}. No TRV_{low} for molybdenum was available, but the toxicity threshold from the literature was a NOEC, thus no further risk characterization for molybdenum is necessary.

- Nickel EPCs in sediment resulted in HQs greater than 1 relative to the TRV_{low} at Pole Canyon Creek (HQ = 12). Only one drainage, Pole Canyon Creek (HQ = 6) had an HQ greater than 1 for nickel when using the TRV_{high}. Nickel in Pole Canyon Creek sediments is carried forward for further risk characterization.

- Selenium EPCs resulted in HQs greater than 1 relative to the TRV_{low} in Pole Canyon Creek (HQ = 17), North Fork Sage Creek (HQ = 3), Hoopes Spring (HQ = 5), and Lower Sage Creek (HQ = 4). The same drainages exceeded the TRV_{high}: Pole Canyon Creek (HQ = 8), North Fork Sage Creek (HQ = 2), Hoopes Spring (HQ = 3), and Lower Sage Creek (HQ = 2). All other drainages had HQs less than 1. Selenium in sediments from the above identified drainages is carried forward for further risk characterization.

- Silver EPCs in all drainages except at Pole Canyon Creek (HQ = 4) resulted in HQs less than 1 relative to the TRV_{low}. Compared to the TRV_{high}, all silver concentrations resulted in HQs less than 1. Silver in sediments from Pole Canyon Creek is carried forward for further risk characterization.

- Thallium EPCs in all drainages resulted in HQs of 1 or less when compared to the TRV_{high}. No TRV_{low} was available for thallium in sediments; however, the threshold derived from the literature was a sediment value resulting in 2 percent of the 25^{th} percent lethal body concentration for *Hyallela azteca*. This sediment value is expected to be sufficiently low; sediment concentrations below this level are expected to produce negligible risks. No further risk characterization for thallium in sediments is warranted.

- Zinc EPCs resulted in HQs less than 1 relative to the TRV_{low} in all drainages except the Pole Canyon Creek drainage (HQ = 11). The Pole Canyon Creek EPC also resulted in an HQ greater than 1 (HQ = 3) relative to the TRV_{high} while all other drainage EPCs were less than 1. Zinc in Pole Canyon Creek sediments is carried forward for further risk characterization.

From the Tier 2 assessment of sediment ECOPCs by drainage, the following ECOPCs and drainages are carried forward into the Tier 3 assessment:

- Arsenic – Pole Canyon Creek;
- Barium – All drainages;
- Beryllium – All drainages;
- Cadmium – All drainages;
- Chromium – Pole Canyon Creek and Sage Creek;
- Copper – Pole Canyon Creek;
- Manganese – Smoky Creek, Tygee Creek, Pole Canyon Creek, North Fork Sage Creek, Sage Creek, South Fork Sage Creek, and Lower Sage Valley;
- Nickel – Sage Creek and Pole Canyon Creek;
- Selenium – Pole Canyon Creek, North Fork Sage Creek, Hoopes Spring, Lower Sage Valley;
- Silver – Pole Canyon Creek;
- Vanadium – All drainages; and
- Zinc – Pole Canyon Creek.

3.3.1.3 Biota

The Site-wide screening level assessment for fish tissues compared the maximum fish tissue concentration to the lowest TRV for each chemical. The following ECOPCs were identified: aluminum, barium, cadmium, chromium, cobalt, copper, iron, manganese, nickel, selenium, vanadium, and zinc. Of these ECOPCs, no TRVs were identified for barium, cobalt, and manganese.

Fish Tissue

Table 3-19 presents the Site-wide Tier 1 assessment for fish tissue which are divided into two groups: trout and sculpins. The Tier 1 assessment indicates only aluminum, iron, and selenium as ECOPCs for trout but no definitive ECOPCs are defined for sculpin, although several are identified as uncertain based on this assessment. The Tier 1 assessment is based on Site-wide data and is intended to reflect conditions affecting populations across all habitats on the Site. ECOPCs identified from the screening-level assessment will be carried forward to the drainage level assessment in Tier 2 even though they might not be identified as ECOPCs from the Tier 1 assessment.

For Tier 2, both trout and sculpin tissue EPCs were assessed for those ECOPCs with defined TRVs (Table 3-20). Trout-specific TRVs were identified for most ECOPCs (exceptions include barium, beryllium, boron, cobalt, manganese, and thallium). Sculpin-specific TRVs were not available. To evaluate sculpin, a surrogate TRV from a sensitive non-trout species was identified from the available data for each parameter. For aluminum and iron, only a trout TRV was identified, thus the sculpin EPCs were compared to the trout TRV. Collectively, for both species no TRVs were identified for barium, cobalt, and manganese.

The tissue EPCs for drainage-specific analyses are based on drainages from which fish tissue data are available. No fish were collected from Roberts Creek, North Fork Sage Creek, or Pole Canyon Creek. The following analysis represents drainages from which trout and/or sculpin data were available from one or more sampling locations.

Results of the Tier 2 drainage-based fish tissue risk characterization are as follows:

- Trout tissues concentrations of aluminum across all drainages exceeded the trout TRV with HQs ranging from 2 to 12.4. Sculpin tissue, where collected, resulted in EPCs that exceeded the trout TRV across all the drainages with HQs ranging from 3 to 12. The sculpin tissue EPC for Hoopes Spring resulted in an HQ of 1.

- Cadmium EPCs for trout tissues from all drainages resulted in HQs of 1 or less. Likewise, all sculpin tissue EPCs were less than the respective TRV with all HQs 0.5 or less.

- Chromium EPCs for trout tissues from all drainages resulted in HQs of 0.8 or less. Likewise, all sculpin EPCs were less than the respective TRV used with all HQs 0.009 or less.

- Copper EPCs for trout tissues resulted in an HQ of 2 for Crow Creek, while all other drainage EPCs for trout tissues had HQs of 1 or less. Sculpin EPCs were all less than the tissue TRV with all HQs 0.2 or less.

- Iron EPCs for trout and sculpin tissues were all greater than the iron trout TRV (no TRV was found for iron in other species). Trout HQs ranged from 3 to 9, while sculpins HQs ranged from 2 to 9. Smoky Creek and Tygee Creek had the highest tissue HQs.

- Nickel EPCs for trout tissues from all drainages resulted in HQs of 0.4 or less. Likewise, all sculpin EPCs were less than the respective TRV with all HQs 0.5 or less.

- Selenium trout tissue EPCs resulted in HQs of 1 or less for all drainages except Hoopes Spring which had an HQ of 2. Sculpin tissue EPCs resulted in HQs of 1 or less for all drainages. If the lower TRV for trout is considered, trout and sculpin tissue HQs for Lower Sage Valley and South Fork Sage Creek would also likely have HQs greater than 1.

- Vanadium EPCs for trout tissues from all drainages resulted in HQs of 0.7 or less. Likewise all sculpin EPCs were less than the respective TRV with all HQs 0.7 or less.

- Zinc tissue EPCs for trout resulted in HQs ranging from 4 to 7. All sculpin tissue EPCs for zinc were less than the respective TRV resulting in HQs of 0.7 or less.

Based on the Tier 2 risk characterization of fish tissues, the following drainages and ECOPCs are carried forward for Tier 3 risk characterization:

- Aluminum – All drainages;

- Copper – Crow Creek;

- Iron – All drainages;

- Selenium – Hoopes Spring, South Fork Sage Creek, and Lower Sage Valley; and

- Zinc – All drainages.

<u>Amphibians</u>

Because of relatively sparse data available on environmental toxicity of metals, amphibians were not included in the BPF (Formation 2013) as a specific receptor group to be evaluated quantitatively in the SSERA. However, at the request of the USFWS via USFS, a literature search was conducted in 2014 to evaluate the sensitivity of amphibians compared to fish and other aquatic life that are being evaluated quantitatively in the SSERA. The available information on the relative sensitivity of amphibians compared to fish toxicity for the same chemicals suggests that amphibians have sensitivity similar to fish (Kerby et al. 2010, Weltje et al. 2012).

Additional literature searches were conducted to obtain information relative to developing a selenium TRV for amphibians. Two TRV for amphibians were identified and described in Section 3.2.1.3. Data from sculpin tissues, where available, were compared to the amphibian TRVs compiled to assess potential risks. Sculpin EPCs for individual sites were used for comparison as shown in Table 3-21.

Based on the sculpin tissue EPCs for sites where sculpin data were available, all HQs based on the NOEC TRV were 1 or less, and all HQs based on the LOEC TRV were less than 1. Based on this result, risk to amphibians is acceptable.

3.3.2 Tier 3 Baseline Assessment

The Tier 3 Assessment involves evaluating ECOPCs on a more location-specific basis to focus the analysis on parts of the Site contributing most to potentially unacceptable risk from ECOPCs. Tier 3 also includes less conservative assumptions about exposure and effects utilized in previous tiers.

3.3.2.1 Surface Water

In Tier 2, ECOPCs were assessed by drainage to identify where ECOPCs exceeded their respective TRVs. Table 3-22 shows the upstream and downstream comparisons for the risk characterization (Tier 3) where ECOPCs exceeded their respective TRVs in the Tier 2 assessment. While the Tier 2 assessment narrowed the risk characterization to specific drainages and ECOPCs, Table 3-22 shows all drainages and ECOPCs carried forward from the Tier 2 assessment, primarily to illustrate where ECOPC concentrations are elevated within a specific drainage. The Tier 3 assessment uses Site-specific hardness for hardness-based criteria to derive criteria for each of the drainage basins.

Discussion of the results from the Tier 3 assessment focuses on those drainage/ECOPC combinations where risk was identified (i.e., HQ >1). Results of the Tier 3 assessment are as follows:

- For aluminum, all risk characterization to this point relied on using the 1988 AWQC for aluminum which was developed using minimal species data under low pH conditions. A revised hardness-based aluminum criterion, as described in Section 3.2.1.1, was used in the Tier 3 risk characterization to estimate risks in surface waters using a more up-to-date criterion. The revised criterion integrates hardness and includes data for species tested over a more natural range of pH. Using these criteria, no risks are found for aluminum in Site surface waters from any drainage.

- Tier 3 screening of arsenic, cadmium, nickel, and zinc identified only the downstream sample location (LP-1) in Pole Canyon Creek as one that may present a risk to aquatic receptors due to these ECOPCs ($HQ_{chronic}$ = 2, 33, 5, and 6, respectively). Cadmium and zinc HQs at LP-1 indicate the potential for acute toxicity as well (HQ_{acute} = 13 and 6, respectively).

- Iron EPCs for upstream locations across the different drainages indicate that iron, for the most part, is lower than the $TRV_{chronic}$ and exceeded the $TRV_{chronic}$ in upstream locations only in North Fork Sage Creek (HQ = 2). At downstream locations, iron EPCs in surface water exceeded the $TRV_{chronic}$ in North Fork Sage Creek and Sage Creek with $HQ_{chronic}$ values of 2 at both locations.

- Selenium was not identified as exceeding the $TRV_{chronic}$ value in any of the upstream portions of any of the drainages. In the downstream portions of Pole Canyon Creek, North Fork Sage Creek, Hoopes Spring, South Fork Sage Creek, Lower Sage Creek, and Crow Creek, HQs were greater than 1 ($HQ_{chronic}$ = 1166, 2, 9, 3, 5, and 2, respectively) (Figure 3-1). HQ_{acute} values greater than 1 were calculated for Pole Canyon Creek and Hoopes Spring (292 and 2, respectively).

Arsenic, Cadmium, Nickel, and Zinc

Arsenic, cadmium, nickel, and zinc in surface waters may pose a risk to aquatic receptors in Pole Canyon Creek downstream of the ODA. For Pole Canyon Creek, risks to aquatic receptors are entirely attributed to those surface waters found at the LP-1 location. As described in the Final RI Report (Formation 2014), with the implementation of the Pole Canyon bypass pipeline, the LP-1 site is now a seep at the toe of the ODA, although historically it was a part of the Pole Canyon Creek stream channel. The bypass pipeline outlet discharges downstream of the LP-1 location at LP-PD. With operation of the bypass pipeline and infiltration basin, pipeline water delivered to downstream Pole Canyon Creek has low ECOPC concentrations which pose no risk to aquatic receptors.

During most years, Pole Canyon Creek flows rarely reach North Fork Sage Creek, as flows are dispersed and infiltrate into Sage Valley soils. During particularly high runoff years, Pole Canyon Creek flows can reach North Fork Sage Creek as shown by several instances of increased selenium concentrations measured in North Fork Sage Creek. While high runoff years may produce sufficient flows to reach North Fork Sage Creek, the surface water data show that increased selenium and potentially other ECOPCs are short-duration pulses occurring during some high-flow events with dilution downstream. This connectivity, while ephemeral, does still pose a risk to aquatic receptors; however, the risk is short term and very localized to a small section of North Fork Sage Creek.

Iron

Elevated iron in North Fork Sage Creek appears to be isolated to two areas, NSV-2 (a spring) and NSV-6 (the downstream-most sampling location on North Fork Sage Creek above the confluence with Sage Creek). Iron in the NSV-2 spring does not appear to reach the downstream portions of North Fork Sage Creek as all iron concentrations are less than 1 mg/l. At NSV-6, iron concentrations do not appear to be due to upstream sources as all iron concentrations are substantially less than 1 mg/L.

In Sage Creek, iron exceeded the criterion once, at the SV-1 irrigation ditch location downgradient of detention basin DP-2. No other Sage Creek sites had iron concentrations higher than about 0.5 mg/l. As discussed below in the sediment characterization, all Sage Creek exceedances for ECOPCs appear to result from the concentrations at this location.

In South Fork Sage Creek, iron was elevated in three samples from 2010 at the LSS-M1 and LSS-M2 springs resulting in an elevated iron concentration downstream at the LSS location. This iron pulse is likely associated with higher flows observed in June. Elevated iron concentrations were not detected at the LSS location during lower flows in November 2011.

Selenium

Selenium concentrations in surface waters of the streams adjacent to the Site are variable; source areas within drainages affect selenium concentrations between upstream and downstream sampling locations relative to these inputs (Table 3-22). Concentrations in Smoky Creek, Tygee Creek, and Sage Creek upstream of the confluence with North Fork Sage Creek were low resulting in HQs less than 1. Pole Canyon Creek, North Fork Sage Creek, Hoopes Spring, South Fork Sage Creek, Lower Sage Creek, and Crow Creek all had HQs greater than 1 for selenium. Other than Pole Canyon Creek and Hoopes Springs where the chronic selenium HQs were 1166 and 9, respectively; chronic HQs for all other drainages were 5 or less. A refined Tier 3 assessment was conducted to evaluate location-specific selenium concentrations as discussed below (Table 3-23).

In Pole Canyon Creek, the selenium HQ was high ($HQ_{chronic}$ = 1166 and HQ_{acute} = 292), due to selenium concentrations at the LP-1 location (a seep at the toe of the Pole Canyon ODA). At the LP-PD location, downgradient from LP-1 and downstream of the Pole Canyon Creek diversion, selenium concentrations were low ($HQ_{chronic}$ less than 1). At the time of the data collection for the RI, there was no surface water connectively from the LP-1 site to the LP-PD site where the diversion water was discharged. Seepage water, however, does persist below the ODA at LP-1 and selenium concentrations in that surface water are high. To pose a risk to receptors, concentrations must be high enough to cause an effect and receptors must be present in order to be exposed. The risk conclusions must consider whether habitat is absent, or water presence is too infrequent for aquatic organisms to live there. Due to the isolated nature of LP-1 and its lack of connectivity to the main stream the majority of the time, exposure to these concentrations is extremely limited for aquatic ecological receptors. However, if aquatic invertebrates are present in sufficient quantity at LP-1, then potential risks to aquatic-feeding receptors could be higher than for LP-PD.

For North Fork Sage Creek, the $HQ_{chronic}$ was 2 based on surface water samples collected at NSV-6 and slightly downstream from this location. The surface water risks (HQs) for other locations in North Fork Sage Creek were all less than 1; the $HQ_{chronic}$ for upstream locations was 0.1. NSV-6 is located downstream of the confluence between Pole Canyon Creek and North Fork Sage Creek. While surface flows in Pole Canyon Creek rarely extend to North Fork Sage Creek, some alluvial groundwater from Sage Valley may discharge to North Fork Sage Creek. These alluvial subsurface discharges would be affected by shallow alluvial groundwater flow from Pole Canyon.

Hoopes Spring is the largest contributor of flow with elevated selenium concentrations to Sage Creek (Hoopes Spring $HQ_{chronic}$ = 9 from Table 3-22). A breakdown of the Hoopes Spring sample collection sites (Table 3-23) shows that the $HQ_{chronic}$ at the main spring site (HS with the highest discharge volume) was 10 while the HQ_{acute} was 3. At the lower discharge volume sites, HS-A1, HS-A2, and HS-C1, $HQ_{chronic}$ values were 3, 3 and 11, and the HQ_{acute} values were less than 1, less than 1, and 3, respectively. In the main discharge channel for Hoopes Spring at HS-2 and HS-3 (near the confluence of Hoopes Spring channel with Sage Creek), the $HQ_{chronic}$ values for selenium were 8 each and the HQ_{acute} values were 2 each. The Hoopes Spring channel provides adequate aquatic habitat, and aquatic receptors including trout and invertebrates are present. Therefore, selenium concentrations in Hoopes Spring represent a potential risk to receptors in the channel and contribute to exposure downstream in Sage Creek.

In South Fork Sage Creek, the perennial stream originates from a series of springs. LSS-M1 and LSS-M2 locations are near the upper end of the perennial channel and had selenium $HQ_{chronic}$ values of 1. The LSS location on South Fork Sage Creek represents a cumulative integration of the various spring discharges, and the selenium $HQ_{chronic}$ value was 3. Farther downstream at location LSS-2, the selenium $HQ_{chronic}$ value was also 3. No HQ_{acute} values were greater than 1 for South Fork Sage Creek. While these springs also discharge from the Wells Formation, the selenium concentrations are substantially lower than concentrations observed in Hoopes Spring. Because South Fork Sage Creek is perennial downstream of the springs, adequate habitat exists and potential exposure occurs and, therefore, potential risk due to selenium concentrations may also be present.

The Lower Sage Valley locations are a series of sites downstream of the confluence of North Fork Sage Creek and Sage Creek. Some are located upstream of the Hoopes Spring channel inflow (see Table 3-9). As shown in Table 3-22, selenium concentrations upstream of the Hoopes Spring input are low and the resulting $HQ_{chronic}$ was less than 1. Table 3-23 shows the HQs for the Lower Sage Valley locations downstream of the Hoopes Spring inflow. The $HQ_{chronic}$ values from upstream to downstream in Lower Sage Creek generally decrease from LSV-1B to LSV-3a (5 locations) ranging from 6 and 7 to 4 reflecting decreasing selenium concentrations. However, at the Sage Creek location farthest downstream from Hoopes Spring, but upstream of the confluence with Crow Creek, the $HQ_{chronic}$ increased to 7. For all of the Sage Creek locations downstream of Hoopes Spring, HQ_{acute} values ranged from 1 to 2 with no consistent pattern in concentrations. It is important to note that downstream of LSV-2C, South Fork Sage Creek discharges to Sage Creek. Locations LSV-3, LSV-3a, and LSV-4 capture the changes in water quality that may occur due to the additional source input from South Fork Sage Creek springs (via South Fork Sage Creek). Throughout its length, Sage Creek downstream of Hoopes Spring has elevated selenium concentrations and adequate habitat for aquatic ecological receptors; therefore, potential for exposure and for risk is present.

Sage Creek discharges to Crow Creek. Selenium concentrations in Sage Creek upstream of Crow Creek are elevated as reflected by the HQ values (Table 3-23). Downstream of Sage Creek, Crow Creek $HQ_{chronic}$ values were 2 each at locations below the confluence and near the Idaho-Wyoming state line. While selenium concentrations in Crow Creek are significantly diluted compared to Sage Creek, concentrations are still about two times the state standard (0.005 mg/L). Adequate habitat for aquatic ecological receptors is present in Crow Creek, receptors are present, and potential for exposure and risk is present based on comparison to the state standard.

In surface waters of the Site, selenium concentrations at locations downstream of significant source areas may pose a risk to aquatic ecological receptors. These estimates of risk, based on HQs, use the existing state standard value of 0.005 mg/L. As noted previously, the state of the science for selenium effects in aquatic organisms indicates that selenium concentrations in egg/ovary tissues provide the best measure of effects in fish, which are sensitive aquatic receptors. While the surface water data suggest risks are present at numerous sites due to aqueous selenium concentrations, the fish tissue data presented in Section 3.3.2.3 will be used to identify more accurate estimates of potential risks at these locations.

3.3.2.2 Sediment

Table 3-24 shows the sediment HQs by site for those drainages identified to have ECOPCs with HQs greater than 1 from Table 3-18.

Arsenic

The only exceedances of the TRV_{low} for arsenic occurred in Pole Canyon Creek, in an ephemeral/intermittent stretch of stream downstream of LP-PD. The low HQ values (HQ = 2), coupled with no exceedance of the TRV_{high} suggest that arsenic in sediments falls within that uncertain area of no and likely probable effects based on the TRVs. Limited risk at these locations is expected because the exposure pathway is limited and likely incomplete due to lack of flows at these locations.

Barium

Barium was carried forward because it exceeded the TRV_{low} and TRV_{high} values at nearly every location. However, the sediment TRVs for barium are not based on toxicity to aquatic invertebrates. In fact there is a paucity of information on the toxicity of barium to freshwater invertebrates and vertebrates because it is nearly insoluble at natural water pH values. Throughout the Site, concentrations of barium in sediments ranged from 52 to 212 mg/kg. In upper Sage Creek and upper Crow Creek above mining influences, barium concentrations in sediments ranged from 61 to 141 mg/kg, which is greater than the TRV_{high} value. The presence of elevated barium concentrations at nearly every location except one of the primary source locations, and its elevated concentrations at locations not influenced by mining at this Site, suggests barium in sediments does not pose a risk due to activities from this Site. No further evaluation of barium in sediments is warranted.

Cadmium

Cadmium in sediments from each drainage was elevated relative to the TRV_{low} value, except at Smoky Creek (USm) and upstream Sage Creek (US and US-4). Where cadmium exceeded the TRV_{low} but did not exceed the TRV_{high}, there is uncertainty about whether or not cadmium poses significant risk to benthic receptors. Cadmium only exceeded the TRV_{high} value in Pole Canyon Creek and in the irrigation ditch related to Sage Creek (SV-1). In these drainages, HQs that exceeded the TRV_{high} value ranged from 3 to 9 (Table 3-24). At these locations, cadmium may pose a risk to benthic invertebrates where/when the exposure pathway is complete. Exposure at the SV-1 location may be limited because this site is in an irrigation ditch downgradient of detention basin DP-2 and it is unclear if the ditch is still used to irrigate in Sage Valley. If not, then no permanent/significant habitat is likely at this location. The magnitude of exposure there is not indicative of the magnitude of exposure in Sage Creek proper as HQ_{high} values in the mainstem of Sage Creek were less than 1. Consistent exposure to elevated cadmium in sediments at Pole Canyon Creek only occurs in the channel where flowing water occurs (LP-PD and downstream). Remnant sediments with elevated cadmium are likely present as cadmium is well below effects thresholds in surface water discharged at this location. At the locations downstream of LP-PD, which are in an ephemeral/intermittent portion of the drainage, exposure is limited due to lack of flows.

Chromium

Only lower Pole Canyon Creek sediments exceeded the TRV_{high} with HQs of 3 for two of the three downstream locations. Chromium at the LP-PD locations exceeded the TRV_{low} (HQ = 2) but not the TRV_{high} (HQ = 0.8). Of the Sage Creek locations, only the SV-1 location had a chromium concentration that exceeded the TRV_{low} (HQ = 2). Similar to the discussion provided above for other ECOPCs, chromium concentrations may exceed TRVs at the downstream Pole Canyon Creek locations, but since permanent aquatic habitat is limited or absent, no adverse effects on aquatic populations is likely due to the lack of exposure. At the Pole Canyon Creek and Sage Creek locations where chromium exceeded the TRV_{low} but not the TRV_{high}, risk may be present, but it is uncertain.

Copper

Copper only exceeded the TRV_{low} value but not the TRV_{high} value in lower Pole Canyon Creek sediments where exposure is limited (HQ =2).

Manganese

North Fork Sage Creek was the only drainage with a manganese concentration that exceeded the TRV_{high} which resulted in an HQ of 5 for location NSV-6. The source of manganese here is unknown given the only potential surface water flows that occasionally reach North Fork Sage Creek are from Pole Canyon Creek. Pole Canyon Creek sediment manganese concentrations exceeded the TRV_{low} value only at LP-PD and the downstream-most sediment location LPT3-2 (HQ = 3). In Smoky Creek, Roberts Creek, Tygee Creek, Sage Creek, and South Fork Sage Creek, manganese in sediments exceeded the TRV_{low} value but not the TRV_{high} value. As evidenced by concentrations at Sage Creek locations upstream of mining activity in Sage Creek (US and US-4), manganese may be a background contaminant, although some uncertainty exists for those concentrations that exceeded the TRV_{low} but not the TRV_{high}.

Nickel

Only Pole Canyon Creek sediments exceeded the Tier 2 screening TRV with HQs that ranged from 2 to 6. Nickel in sediments, primarily at LP-PD, likely poses a risk to benthic receptors as the pathway for exposure is complete at that location, but is partially if not fully incomplete at the lower Pole Canyon Creek sediment locations (including the creek channel in northern Sage Valley). In Sage Creek at SV-1, nickel in sediments exceeded the Tier 1 TRV but not the Tier 2 TRV while nickel in the remaining locations for Sage Creek was less than the Tier 1 TRV.

Selenium

Selenium in sediments from Hoopes Spring and North Fork Sage Creek exceeded the TRV_{high} with HQs of 3 and 2, respectively. In Pole Canyon Creek, sediments were collected from 11 locations; for 10 of those locations two depth intervals were collected (0-4 inches and 6-12 inches) within the Pole Canyon Creek flow path into Sage Valley, although the bulk of those sediments were collected from a dry stream bed. At the LP-PD location where water was present, the selenium concentration in sediments was in the lower part of the range (13.4 mg/kg) given selenium near the terminus of the flow path was measured at 96.7 mg/kg. Pole Canyon Creek HQs for selenium in sediments, relative to the TRV_{high} value, ranged from 2 to 12 and may pose a risk to higher trophic level consumers. In Lower Sage Valley, selenium in sediments exceeded the TRV_{high} value at LSV-2C, LSV-3, and LSV-4 locations. Risk to higher trophic level consumers may be present in Lower Sage Valley. Recall that for selenium, however, the TRVs are not based on effects to benthic organisms, but rather as potential bioaccumulation effects to organisms that consume those benthic organisms.

Silver

Only lower Pole Canyon Creek sediments exceeded the TRV_{low} value with HQs that ranged from 3 to 4. At these locations the pathway is likely incomplete as described previously.

Zinc

Only lower Pole Canyon Creek sediments exceeded the TRV_{high} value with HQs of 3 each. Zinc in sediments from all Pole Canyon Creek locations exceeded the TRV_{low} (HQs ranged from 4 to 11). As mentioned previously, exposure at most of these locations is limited, thus potential risk is also limited. Only at the LP-PD location is exposure present and likely consistent. Risk to benthic invertebrates due to zinc in sediments at LP-PD is uncertain.

Metal Mixtures

Because sediments may contain a mixture of metals, it is often necessary to assess the potential cumulative toxicity of several metals. MacDonald et al. (2000) proposed that the mean PEC–Q was a useful tool to estimate cumulative metal mixtures in sediments, based on analysis of hundreds of field collected sediment samples that were known to be either toxic or non-toxic. The mean probable effects concentration quotient (PEC-Q) is the average of each PEC quotient (metal concentration in sediment divided by its respective PEC). Through evaluation of a large quantity of sediment samples, MacDonald et al. (2000) were able to determine a metric which accounted for the highest predictive ability of toxic sediments. Various forms of the PEC-Q ratio as a measure of cumulative risk have been cited by several credible sources. MacDonald et al. (2000) used a value of mean PEC-Q<0.5; Florida inland guidance (MacDonald and USGS 2003) suggested a cleanup sediment guideline of 0.6. Tri-States

(MacDonald et al. 2009) suggested mean PEC-Q values from 0.556 to 1.11 depending on how metals were grouped. Sediment risk benchmarks for cumulative risks at the Portland Harbor Superfund Site are based on a mean PEC-Q of 0.7[3].

Because of this range, it is appropriate to evaluate mean PEC-Q values against a range of cumulative thresholds. For those parameters with established PEC values, sediments with a mean PEC-Q of greater than either 0.5 or 0.7 are considered to be mostly toxic, while sediments with a mean PEC-Q less than 0.5 or 0.7 are likely not toxic. Table 3-25 shows the mean PEC-Q values for those parameters with established PEC values for sediments from each of the drainages evaluated. Only the Pole Canyon Creek drainage sediments were found to have a mean PEC-Q greater than 0.5 and 0.7. In Sage Creek, the mean PEC-Q exceeded the 0.5 threshold but not the 0.7 threshold. Cadmium was driving the potential cumulative risk for this drainage based on elevated cadmium levels at a single irrigation ditch site (SV-1). Removal of the higher cadmium concentration for SV-1 would result in a PEC-Q for cadmium in Sage Creek of 0.52. Recalculating the mean PEC-Q using this value results in a value of 0.38 which is below the lower cumulative metal potential risk threshold of 0.5. Note that selenium is not part of this assessment as there are no toxicity-based benchmarks for selenium for aquatic invertebrates.

Additional Metals

Using the logic described above, additional metals detected during the RI were also included. These additional metals had TRVs developed using different approaches than those used to develop the PEC values. Table 3-26 shows the addition of these metals in the mean PEC-Q process. The results are similar to those noted above; only Pole Canyon Creek sediments yielded a mean PEC-Q greater than 0.7. North Fork Sage Creek had a mean PEC-Q greater than 0.5 due to high manganese at the NSV-6 location. Similarly, the Sage Creek mean PEC-Q was also greater than 0.5 due to the issue of a single cadmium concentration at SV-1, described previously. Based on these data, risks to aquatic benthic macroinvertebrates due to concentrations of multiple ECOPCs is likely in Pole Canyon Creek where the pathway is complete. Potential cumulative risks may also be present at some localized areas such as Sage Creek (SV-1) and NSV-6 if the pathway is complete for exposure.

With the exception of barium and probably manganese, all other sediment ECOPCs are associated with known source areas, predominantly Pole Canyon Creek. However, exposure is limited because most of the sediments evaluated for Pole Canyon Creek were collected from a dry stream bed. The concern for these sediments would be transport during high flow years if flows are significant enough to mobilize sediments for transport downgradient to North Fork Sage Creek.

[3] Reference to establish the value of 0.7 based on results for the Willamette River at the Portland Harbor Superfund Site (Windward Environmental 2013).

Additional LOEs – Benthic Invertebrate Community Metrics and Tissue Concentrations

Risks to benthic invertebrates due to selenium in sediments cannot be evaluated using traditional benchmarks because no traditional benchmarks are available. In this subsection, benthic tissue concentrations are compared to benthic tissue thresholds developed from the literature as described previously. Table 3-27 shows the maximum benthic invertebrate tissue concentration for each location where samples were collected. These measured concentrations are compared to the two benthic invertebrate TRVs derived from the literature. As shown, only two locations in Sage Creek (LSV-2C and LSV-4) have benthic invertebrate tissue concentrations that exceed the NOEC and LOEC tissue TRVs. All other locations across the Site have HQs of 1 or less. This result is not surprising given that the preponderance of the literature suggests that benthic invertebrates are not particularly sensitive to selenium, and that fish are more sensitive receptors. While the previously presented risk characterizations for selenium in sediments suggested a number of locations where selenium is a risk (to higher order receptors due to sediment concentrations), the current evidence indicates that selenium in sediments for the most part is not a risk issue to benthic macroinvertebrates. Risk to higher trophic level consumers for this SSERA is evaluated through the evaluation of fish tissues.

Another line of evidence to consider for selenium in sediments is the components that make up community structure and function. The Stream Macroinvertebrate Index (SMI) is compared through time at three different locations along with benthic invertebrate tissue concentrations to evaluate if any temporal trends are obvious relative to selenium concentrations. This represents a qualitative examination of available data to provide another LOE to determine whether conditions have impacted the benthic community at the Site.

Similar to the Index of Biotic Integrity (IBI) which is a multi-metric index of the structural and functional condition of a benthic macroinvertebrate community, Idaho uses an index called the Stream Macroinvertebrate Index (SMI). IDEQ's *Small Stream Ecological Assessment Framework: An Integrated Approach* (Grafe 2002) provides documentation and methods for deriving an SMI. The following nine metrics are calculated for the benthic community samples in order to derive the SMI score: total taxa, trichoptera taxa, percent 5 dominant taxa, ephemeroptera taxa, percent Plecoptera, scraper taxa, plecoptera taxa, HBI, and clinger taxa. From 2006 to 2008, Simplot collected information about aquatic communities at various locations around the mine to support development of a Site-specific selenium criterion. In 2009, similar overlapping sites were also monitored as part of a mitigation and monitoring plan in advance of additional mining south of Panel E. These data were collected under Agency-approved work plans. Briefly, three separate surber samples were collected and composited for benthic community enumeration and identification. SMI metrics were derived from the samples collected from each location. Because of the potential seasonal biases that may occur, SMI data were only developed during late summer/fall periods.

Figure 3-2 illustrates the SMI scores for three sites from the 2006 to 2011 fall sampling periods. The three sites selected represent different levels of exposure within the streams evaluated, including Crow Creek upstream of Sage Creek (CC-350), Sage Creek downstream of Hoopes Spring (LSV-2C), and Crow Creek downstream of Sage Creek (CC-1A). Qualitatively, there does not appear to be discernable downward trends in the SMI (lines on Figure 3-2), particularly at the LSV-2C site which has the highest overall aqueous selenium exposure. Benthic macroinvertebrate tissue concentrations were compiled for each of these locations and plotted for each year data were available (bars on Figure 3-2). For the most part, benthic invertebrate tissue concentrations for Crow Creek upstream of Sage Creek and Crow Creek downstream of Sage Creek are well below the benthic tissue thresholds for mayflies (discussed in more detail below). However, benthic tissue thresholds for Sage Creek just downstream of Hoopes Spring exceed these thresholds.

Many factors can influence benthic macroinvertebrate structure and function such as quality and quantity of physical habitat, fish predation, seasonal temperature extremes, and grazing pressures that affect sedimentation, among other factors. The proximity of these locations to one another (despite their size differences), suggest that these three locations experience similar flow changes, predation by fish, temperature extremes, and grazing pressures, and have moderately good habitat quality. The major difference between each of these sites, however, is the aqueous selenium concentrations, which in turn affects the selenium concentrations in benthic invertebrates. If selenium concentrations rise above an effect threshold, adverse impacts to a metric such as the SMI should be observed. The six-year SMI scores for each site suggest a slightly downward trend from 2006 to 2011, despite the upward trend in benthic invertebrate selenium tissue concentrations at LSV-2C and CC-1A. If selenium toxicity to aquatic benthos was indeed occurring at the environmental concentrations observed, particularly in LSV-2C where an increase in aqueous selenium concentrations of more than 0.02 mg/L occurred over four years along with an approximately 35 mg/kg dw increase in benthic tissue concentrations, the SMI metrics might reflect the increased toxicity. Of course, other factors may play a more important role in benthic community structure and function than selenium toxicity, but one might expect that a change of this magnitude would be discernable by the SMI metric. Conversely, the SMI metric may not discern such a change if the benthic tissue thresholds are overly conservative and not representative of potential effects of selenium on benthic invertebrates at this Site.

3.3.2.3 Biota

Fish Tissue

The primary biotic media sampled for the RI for aquatic resources was fish tissue. In the Tier 2 baseline assessment, fish tissue concentrations of ECOPCs were evaluated by drainage; aluminum, copper, iron, selenium and zinc may pose a risk to fish based on accumulation of those ECOPCs in fish tissue. No fish tissue TRVs were available for barium, cobalt, and manganese. Table 3-28 lists the trout tissue EPCs for each ECOPC by drainage and HQs calculated for available data. Similarly, Table 3-29 lists the sculpin tissue EPCs for each ECOPC by drainage and HQs calculated for available data. Table 3-30 lists a range of possible trout tissue background ECOPC concentrations that will need to be considered for some ECOPCs found Site-wide.

Evaluation of data from other sites and guidance from other western states suggests that concentrations of some metals in surface water at the Site are within natural background ranges. Background is being evaluated for other phosphate mining sites in southeastern Idaho and may provide information relevant to the Smoky Canyon Mine. However, conclusions regarding background concentrations cannot be established until agreement on Site-specific background has been reached.

Aluminum

Concentrations of aluminum in trout tissue were highly variable, but both species tended to have tissue concentrations that exceeded the tissue TRV. For some sites, such as US, US-4, and CC-350, which are upgradient of mine influences, the tissue TRV was exceeded by many times the TRV value (HQ range of 3 to 5). Sculpin tissue concentrations, similar to trout, exceeded the TRV at all locations except Hoopes Spring, where HQs ranged from 3 to 12. It appears that aluminum is rather ubiquitous throughout the Site based on its presence in surface waters.

Copper

Copper in trout tissues exceeded the tissue TRV in Sage Creek, Lower Sage Creek, and Crow Creek in both YCT and brown trout. In Sage Creek, copper HQs ranged from 0.8 to 2 at sites upstream and downstream of the haul road in YCT. In lower Sage Creek downstream of Hoopes Spring, copper HQs ranged from 0.6 to 2 in brown trout and YCT, respectively. In Crow Creek, upstream of Sage Creek, the copper HQ was 2. Copper concentrations in sculpins do not appear to represent toxic levels since all HQs were less than 1; however, there is uncertainty in using a tissue TRV for another species. Low copper HQs are, however, more a function of low copper concentrations in sculpin tissues (as compared to trout tissues) rather than the TRV. Copper is an essential micronutrient, and thus will be accumulated to some level in fish tissues. Whether or not the levels observed are toxic is questionable and is explored further in the uncertainty discussion (Section 5).

Iron

Iron is an essential micronutrient, but at some level above nutritional requirements, it may be toxic. Iron in nearly all trout tissue samples for both species resulted in HQs greater than 1 with HQs that ranged from 2 to 9. Iron in sculpin tissues was similarly elevated; all HQs for iron in sculpin tissues exceeded 1. The TRV used to assess iron bioaccumulation toxicity and the natural levels of iron in the environment pose an uncertainty about the accuracy of the risk estimates. Whether or not the levels observed are toxic is questionable and is discussed further in the Uncertainty Analysis (Section 5).

Selenium

For selenium, the TRVs used for tissue were the egg/ovary effects threshold values (e.g., USEPA-derived 18.09 mg/kg dw and Simplot-derived 20.5 mg/kg dw) translated to whole body tissue concentrations by dividing the egg/ovary concentrations by the whole body tissue translation factor (1.45).[4] Whole body tissue concentrations resulting from this translation are 12.48 mg/kg dw (USEPA-derived) and 14.14 mg/kg dw (Simplot-derived). Brown trout have been identified in USEPA (2015) as one of the more sensitive species. Both the TRVs used are based on brown trout tissue effects data. HQs greater than 1 for trout tissues were identified in Hoopes Spring and lower Sage Creek downstream of Hoopes Spring, with HQs of 2 each (Figure 3-3). In sculpins, selenium in tissues resulted in HQs greater than 1 in lower Sage Creek and Hoopes Spring. These results are relatively consistent with the surface water risk characterization and the relationship of these drainages to source areas.

A comparative selenium risk characterization is presented in Table 3-31. Surface water selenium concentrations for each site(s) are shown as EPCs and compared to the chronic state standard for selenium (0.005 mg/L). If fish tissue data were collected from the site(s), then those whole body tissue data were compared to an egg/ovary effect threshold value that was translated to a whole body value to derive a tissue HQ. Finally, as an additional assessment, the Site-specific tissue criterion value was translated to a selenium water concentration value, using the Presser and Luoma (2010) model as modified by USEPA in the 2015 Draft Criterion, primarily for those sites where no fish were collected, but also applied to each site for comparative purposes.

Selenium in fish tissues collected from Hoopes Spring and Lower Sage Creek indicate a selenium risk to fish species in these creeks based on a whole body fish tissue TRV of 14.14 mg/kg dw.

In South Fork Sage Creek and Crow Creek, the fish tissue HQs equaled 1, again based on a fish tissue TRV of 14.14 mg/kg dw. Selenium risk based on fish tissues is within acceptable ranges in South Fork Sage Creek and in Crow Creek. Further discussion of the uncertainties associated with the selenium risk estimates are provided in the Uncertainty Analysis (Section 5).

Data on brown trout populations (standing crop biomass) were plotted against fish tissue selenium concentrations for 2006 through 2013 data (Figure 3-4). The spread of these data

[4] Both the USEPA-derived and Simplot-derived egg/ovary effect threshold values were used at the request of the Agencies to illustrate the difference between the two calculation approaches.

suggests that risk to brown trout subpopulations, based on biomass, may be affected when whole body tissue concentrations exceed about 24 mg/kg dw. A relatively significant drop in brown trout biomass is observed when whole body tissue concentrations were higher than 24 mg/kg dw (which is higher than the whole body tissue threshold used to assess potential developmental effects). The SSERA is conservatively estimating the potential for adverse effects; however, the goal is always to extrapolate laboratory individual endpoints to population-level effects in the field since the assessment endpoint for this SSERA is based on risks to populations. The field data complement the toxicity study data by providing another LOE about the threshold developed. Field measures of populations have uncertainties; however, the population data for the streams at the Site are of high quality and based on agency-approved sampling plans. The data represent a more direct measure of the assessment endpoint, and incorporate other factors that control populations. If population level declines were observed well below the tissue TRV, then the ECOPC would be considered a prime candidate as the causative factor. In this case, however, Figure 3-4 illustrates that even though tissue concentrations exceed the tissue TRVs used in this SSERA, trout populations have flourished. The sharp drop off in standing crop biomass of brown trout above whole body concentrations of 24 mg/kg dw may be due to selenium, habitat factors, or a combination of both factors which will be evaluated more in the Uncertainty Analysis (Section 5).

Zinc

As for copper and iron, zinc is an essential micronutrient, but at some level above nutritional requirements, it may be toxic. Based on the trout tissue TRV for zinc, HQs for zinc greater than 1 are widespread and occur within tissues collected from all drainages. Zinc trout HQs ranged from 3 to 10. Sculpin tissue concentrations of zinc were typically lower than those found in trout. Furthermore, the TRV used for sculpin was also considerably greater than that used for trout which resulted in all sculpin tissue HQs for zinc of less than 1. If the trout tissue TRV would be used for sculpin, then sculpin HQs would likely be 2 or higher at all locations. The TRV used to assess zinc bioaccumulation toxicity and the natural levels of zinc in the environment pose an uncertainty about the accuracy of the risk estimates which will be discussed further in the Uncertainty Analysis (Section 5).

<u>Amphibians</u>

Data on bioaccumulation and developmental toxicity for metals suggest that at least anuran amphibians do not accumulate substantially higher concentrations of metals than fish, and that tissue-based TRVs for fish are protective of the amphibians. The comparative analysis using sculpin tissue data as a surrogate for amphibian tissues, compared to a no and low effect TRV for amphibians, resulted in no HQs greater than 1 for locations where sculpin tissue data were collected (Table 3-21). If that same approach is taken using fish tissues as a suitable surrogate for amphibians to assess potential risks, then comparison of the trout tissues from two species (e.g., brown trout and YCT, two species with different feeding strategies) to the amphibian TRVs would also be logical to fill in data gaps where sculpins were not found. Table 3-32 presents the results of this comparison using trout tissue data. No HQs greater than 1 were calculated for any location. Therefore, the risk analysis presented for the ECOPCs above, and particularly for selenium, can be considered protective of the amphibians at the Site.

4.0 TERRESTRIAL AND RIPARIAN RISK ANALYSIS AND CHARACTERIZATION

The risk analysis phase of the baseline SSERA is the sixth step in the risk assessment process. Risk analysis includes two steps: exposure analysis and effects analysis. Exposure analysis is used to quantify the degree to which receptors are exposed to ECOPCs in each exposure domain. Effects analysis attempts to determine the relationship between exposure to ECOPCs and observed or potential effects to the assessment endpoints.

The estimation of exposure, effects, and the characterization of risk for terrestrial and riparian receptors are discussed in detail in the following sections. The assessment and measurement endpoints associated with the potentially exposed receptor groups were discussed in Section 2.7 and form the basis for the exposure assessment and effects assessment to be completed under Step 6 of the USEPA ERA process (USEPA 1997).

Risk characterization (Step 7), presented in Section 4.3, evaluates the results of the risk analysis and provides a comparison of the Site-specific exposures (Section 4.1) to toxicity benchmarks and TRVs (Section 4.2), and provides information on interpreting the results for the assessment endpoints (Figure 1-3).

4.1 Terrestrial and Riparian Exposure Analysis

The exposure estimates prepared for the SSERA provide an assessment of the potential for exposure through the evaluated pathways identified in the ECSM (Figure 2-11). This is performed as a tiered assessment.

For terrestrial and riparian receptors, measures of exposure are defined as those measures that describe the location and concentration of ECOPCs in abiotic and biotic media that are used to estimate exposure to ECOPCs. Exposures were assessed in a tiered approach based on scale, as follows:

- Tier 1 – Site-wide;
- Tier 2 - By reclamation area (and Sage Valley) for terrestrial receptors and by drainage (see below) for aquatic/riparian receptors; and
- Tier 3 – By sampling location.

There were three basic routes of exposure identified in the BPF and quantified in the exposure assessment: (1) ingestion from food, soil/sediment, and surface water; (2) direct contact (absorption); and (3) inhalation.

Quantification of exposure requires data on ECOPC concentrations in Site environmental media (i.e., soil, sediment, surface water and prey items) and estimates of ingestion rates or contact information for each receptor and pathway. Except as specifically noted in the following sections, only measured concentrations of ECOPCs in prey and other exposure media were used to estimate exposure.

In addition, body weights, ingestion rates of food, and other factors must be known for each of the receptors. The exposure information used for each receptor is unchanged from the screening steps provided in Section 2.7 and are provided in Tables 2-14 (feeding habits) and 2-15 (exposure parameters).

The quantification of receptor-specific exposures via inhalation or dermal absorption was not evaluated because of a lack of appropriate exposure and toxicity data. The exposure of animals to contaminants in soil by dermal contact is likely to be small due to barriers of fur, feathers, and epidermis (USEPA 1989). Due to the uncertain nature of assessing inhalation risk to ecological receptors, inhalation of particulate forms of ECOPCs is assumed to be of lower significance as an exposure pathway as ingestion of contaminated materials at the Site. Thus, the exposure analysis focuses on the ingestion pathway as the primary exposure route for terrestrial vertebrates.

The same exposure model as defined in Section 2.7 was also used in the exposure assessment for wildlife receptors. The generic equation used to calculate intake is:

$$\mathrm{Dose}_{\mathrm{Total}} = (\mathrm{SUF}) \times \frac{\left[\left(\mathrm{C}_{\mathrm{media}} \text{ x } \mathrm{IR}_{\mathrm{media}}\right) + \left(\mathrm{C}_{\mathrm{prey}}\right)\left(\mathrm{IR}_{\mathrm{prey}}\right)\right]}{\mathrm{BW}}$$

Where:

$\mathrm{Dose}_{\mathrm{Total}}$= Daily dose resulting from ingestion of abiotic media and dietary items (milligrams chemical per kilogram body weight per day [mg chemical/kg BW/Day]).

$\mathrm{C}_{\mathrm{media}}$ = Concentration of chemical in abiotic media (mg/kg or mg/L) during incidental ingestion of that media.

$\mathrm{C}_{\mathrm{prey}}$ = Measured concentration of chemical in prey or forage types (mg/kg).

IR = Ingestion Rate (the amount of prey items, surface water, sediment, and soil ingested per day) (kg/day, kg/kg BW/day).

BW = Body Weight of receptor species (kg).

SUF = Site Use Factor to account for the amount of time that the organism spends using the Site.

For the initial calculations in each tier, the SUF was assumed to be 1.0 for all receptors.

4.1.1 Tier 1 Exposure Assessment

Tier 1 exposure was defined in the BPF as an estimate of Site-wide exposure. Tier 1 exposure is a general estimate of exposure that the entire population of each receptor inhabiting the Site may experience. For receptors with small home ranges, such as small mammals and songbirds, the Tier 1 estimate represents an overall risk estimate for the part of the local population that resides within the Site boundaries. For wide-ranging receptors such as the coyote, mule deer, and northern harrier, the Site may represent only a portion of the feeding range for individual animals. For these receptors, the Tier 1 estimate provides an estimate of their exposure while feeding at the Site.

Tier 1 exposures were estimated as the 95UCL concentrations for each ECOPC for each of the exposure media. The 95UCL concentrations were calculated using USEPA ProUCL (version 5.0) statistical package and the values recommended by the software were used as Tier 1 EPCs (USEPA 2013). Statistical summaries of each of the Tier 1 ECOPCs, including the calculated 95UCLs, are provided in Appendix B. The 95UCL EPCs used in the Tier 1 exposure assessment for upland wildlife receptors are provided in Table 4-1. Site-wide soil EPCs for those ECOPCs identified for vegetation and invertebrates only are provided in Table 4-2.

Tier 1 EPCs for aquatic media (sediment, fish tissue, macrophytes, and benthic invertebrates) were not calculated. The receptors potentially exposed to aquatic media (belted kingfisher, mallard, raccoon, and mink) are discussed in Tier 3 of the exposure analysis due to the drainage-by-drainage sampling conducted for those media under the RI. Similarly, exposure to the remaining riparian receptors (song sparrow, meadow vole, and red-winged blackbird) is also discussed in detail in the Tier 3 exposure analysis because, due to habitat limitations, they are not expected to be consistently present in the upland portions of the Site.

The detailed results of the Tier 1 exposure calculations are provided in Appendix C. A summary of the exposure calculated for each ECOPC/receptor pair are provided in Table 4-3.

4.1.2 Tier 2 Exposure Assessment

Tier 2 of the exposure assessment estimates exposure separately for the reclaimed areas of the Site and the parts of northern Sage Valley that may have been affected by transport of ECOPCs but where no mining activities have occurred. The Tier 2 analysis was included to allow comparison among the reclamation approaches that have been employed for ODAs at the Site, and to show a separate risk estimate for northern Sage Valley.

Tier 2 exposures were estimated as the 95UCL concentrations for each ECOPC for the following areas (Figure 4-1):

- Panel A Area 1;
- Panel A Area 2;
- Panel D North;
- Panel D South;
- Pole Canyon ODA;
- Panel E Area 1;
- Panel E Area 2; and
- Northern Sage Valley.

Tier 2 exposures were also summarized for terrestrial receptors in each reclamation (cover) type (Figure 4-2):

- No Cover – seeding directly on overburden without the use of topsoil;
- Topsoil Only – surface topsoil (stripped and stockpiled prior to mining an area) placed at various thicknesses, followed by seeding;
- Topsoil Over Chert – surface topsoil placed over a chert layer, followed by seeding; and
- Dinwoody – surface Dinwoody material placed over a chert layer, followed by seeding.

The 95UCL concentrations for each exposure domain were calculated using USEPA ProUCL (version 5.0) statistical package and the values recommended by the software were used as Tier 2 EPCs. Statistical summaries of each of the Tier 2 ECOPCs, including the calculated 95UCLs, are provided in Appendix B. The 95UCL EPCs used in the Tier 2 exposure assessment for upland wildlife receptors are provided in Tables 4-4 and 4-5. Site-wide soil EPCs for those ECOPCs identified for vegetation and invertebrates only are also provided.

The detailed results of the Tier 2 exposure calculations are provided in Appendix C. A summary of the exposure calculated for each ECOPC/receptor pair are provided in Tables 4-6 through 4-17.

4.1.3 Tier 3 Exposure Assessment

The Tier 3 exposure assessment provides an evaluation of ECOPC concentrations for individual sampling locations where soil and other environmental media were collected. The location-specific estimates are not representative of overall exposures for individuals or subpopulations, and are provided only to show the relative contribution from different areas of the Site to overall exposure and risk estimates. The results are intended for use by risk managers to help develop remediation alternatives. These include both upland and riparian locations.

The upland sampling locations for which exposure was estimated are shown in Figure 4-3. Exposure to riparian receptors was not estimated at these locations since riparian habitat is not present. The riparian and seep sampling locations (Figure 4-4) were identified in the RI/FS Work Plan (Formation 2011b) as areas representative of possible ‘worst-case’ exposure areas for terrestrial and semi-aquatic receptors inhabiting the drainages downgradient of the mine panels and ODAs and at several seeps adjacent to the ODAs. Potential exposure in these areas is also estimated in the Tier 3 exposure assessment. Exposure in riparian areas was estimated for both upland and riparian receptors. At riparian/seep areas where fish are not present, exposure to fish-eating receptors was not calculated.

The EPCs and exposure calculations for each sampling location are provided in Appendices B and C, respectively.

4.2 Terrestrial and Riparian Effects Analysis

The exposure of terrestrial and riparian vertebrate receptors, expressed as the daily rate of intake of chemical, was estimated for each ECOPC/receptor pair. Risk characterization was based on comparison of these estimated intakes to TRVs which are intakes with known levels of toxicity for standard test species. TRVs are generally derived from the scientific literature on toxicity of chemicals. Two types of TRVs are used for the wildlife in the terrestrial risk assessment. The NOAELs are intake rates below which no adverse ecotoxicological effects are expected. NOAEL TRVs are typically used in screening-level ERAs to identify COPCs that are clearly not present at ecotoxic levels. Lowest-Observed Adverse Effects Level (LOAEL) TRVs were used in the risk characterization for wildlife. LOAELs represent the lowest exposure levels evaluated in the referenced toxicity study that were associated with adverse effects. The true threshold for effects is between the NOAEL and LOAEL.

TRVs for wildlife exposure were obtained from regulatory guidance and scientific literature. For ECOPCs for which EcoSSL documents are available, TRVs representing the lowest bounded LOAEL for growth, reproduction, and mortality endpoints from the EcoSSL documentation were utilized in the SSERA. In addition, the geometric mean NOAEL of growth and reproduction endpoints, as well as the geometric mean NOAEL of growth, reproduction, and mortality endpoints were also considered in the SSERA.

For some chemicals, LOAELs were obtained from other major ecological risk databases, including Sample et al. (1996), Los Alamos National Laboratory (LANL 2008), and U.S. Army Center for Health Promotion and Preventive Medicine (USACHPPM 2009). Wildlife TRVs presented in the BPF and considered for use in the SSERA are provided in Table 4-18. It should be noted that not all of the TRVs presented in the BPF and in Table 4-18 were used in quantitative risk characterization.

Consistent with USEPA guidance (1997, 1998), TRVs representing a range of effects based on chronic exposures are included to provide risk managers with more context for risk management decisions.

4.3 Terrestrial and Riparian Risk Characterization

Risk characterization represents Step 7 in the eight-step USEPA ERA process and evaluates the results of the risk analysis completed in Step 6. The risk characterization provides a comparison of Site-specific exposures to toxicity benchmarks and TRVs, and provides information on interpreting the results for the assessment endpoints (USEPA 1997).

HQs are a standard approach identified in USEPA guidance (1997) used to make these comparisons. The HQ is a ratio of the estimated exposure concentration to the TRV where:

$$HQ = \text{Intake}/\text{TRV}$$

In general, if the HQ is less than 1.0 for the NOAEL TRV then no adverse effects are predicted. If the HQ for the LOAEL TRV is less than 1.0, adverse effects to receptor populations are considered highly unlikely. There is, however, no clear consensus from either USEPA guidance or the scientific literature concerning the significance of the level of departure from HQ greater than 1.0 (Menzie et al. 1992). If the intake exceeds the LOAEL TRV, the risk of adverse effects is more likely, but useful characterization of the risk may require more explicit evaluation including spatial extent of elevated concentrations, habitat quality, details of the TRV, and location of elevated exposure. For instance, a high HQ that results from a small, isolated area of high concentration and/or low habitat quality may not indicate potential population/ community-level effects because exposure is limited to a few individuals that must utilize other areas to maintain populations. By contrast, if the highest concentrations occur in limited areas of low habitat quality, risk of adverse effects may be small and remediation in such areas may not result in significant risk reduction.

In the following sections, the tiered risk calculations are summarized. HQs were calculated for all ECOPC/receptor pairs first using the NOAEL TRV used in the screening assessment and the lowest LOAEL TRV identified in Table 4-18 for ECOPCs.

Tier 1 and Tier 2 HQs were calculated for the upland receptors. No ECOPC/receptor pairs were removed from further consideration in Tier 2 or Tier 3 based on the results of the Tier 1 assessment. If low or very low risk was estimated for an ECOPC/receptor pair in Tier 2, then

Tier 3 risks were not estimated and risk to that ECOPC/receptor pair was reported as low or very low.

For riparian receptors, Tier 1 risk was not calculated due to the limited riparian and seep exposure data available (1 sample per drainage). Riparian risks were calculated in Tiers 2 and 3 and conclusions were based on the same methods as described for upland receptors. For all ECOPCs, if risks were not identified as low or very low in the Tier 1 and Tier 2 calculations, then the potential risks from that ECOPC are discussed in more detail in the Tier 3 risk characterization.

4.3.1 Tier 1 Risk Characterization

Tier 1 risks were characterized using the Tier 1 Site-wide EPCs presented in Table 4-1 which were compared to both the NOAEL TRVs used in the screening step and the lowest LOAEL TRV provided in Table 4-18. The Tier 1 characterization is intended to represent exposures to populations of receptors occupying the entire Site. Tier 1 assessment also assumes that exposure potential is the same across the Site (i.e., there is no preference for higher quality habitats within the Site).

For receptor species with small feeding ranges (small mammals, songbirds), the Site represents a subpopulation (within the local population) that is exposed almost entirely to conditions at the Site. Smaller subpopulations of these receptors can also be assumed to inhabit the areas discussed in Tier 2. For species with larger ranges, such as ungulates, coyote, and raptors, the Site represents a portion of the feeding area primarily for individuals and/or a small portion of the local population. Home range information is provided in Table 2-17.

For upland wildlife receptors, the HQ calculations are presented in Tables 4-19 and 4-20 for all ECOPC/receptor pairs. Comparisons of the Site-wide soil 95UCL EPCs to the terrestrial plant and invertebrate receptors are provided in Table 4-21.

Based on the Site-wide NOAEL HQs less than 1.0 for all upland wildlife receptors (Table 4-19), Tier 1 risk for the following ECOPCs is considered *de minimus* and, therefore, LOAEL-based HQs were not calculated for Tier 1:

- Antimony;
- Arsenic;
- Cadmium;
- Chromium;
- Manganese; and
- Nickel.

HQs calculated using the NOAEL TRV were greater than 1.0 for:

- Copper – coyote and northern harrier;
- Lead – northern harrier;
- Molybdenum – deer mouse, eastern cottontail, and mule deer;
- Selenium – all receptors;
- Vanadium – American robin and northern bobwhite; and
- Zinc– northern harrier.

Tier 1 HQs using the lowest LOAEL TRV were calculated for the ECOPC/receptor pairs with one or more NOAEL HQs greater than 1.0 in Table 4-20. Of these, only molybdenum had no LOAEL HQs greater than 1.0 for any receptors.

For the terrestrial plant and invertebrate receptors, Tier 1 EPCs were greater than the screening benchmark for either receptor (Table 4-21) for the following ECOPCs:

- Boron;
- Chromium;
- Manganese;
- Mercury;
- Molybdenum;
- Nickel;
- Selenium;
- Vanadium; and
- Zinc.

4.3.2 Tier 2 Risk Characterization

The intent of the Tier 2 assessment is to compare exposure and potential risks among the reclamation types among the mine panels and ODAs. In addition, Tier 2 EPCs separately characterize exposure in riparian habitats along stream reaches with potentially different contaminant sources within the mine area. As a result, the Tier 2 risks are not intended to be representative of risks for the Site-wide populations as addressed in the assessment endpoints (Table 2-21). In the case of receptors with small home ranges (i.e., significantly less than the size of the mine panel), the Tier 2 risks are likely to be representative of risks to the subpopulations of receptors that may inhabit the area in which exposure and risk are estimated. For receptors with moderately large home ranges (i.e., similar in size to the mine panel), the Tier 2 risks are representative of risks to an individual. For receptors with large home ranges (i.e., larger than the area of assessment), risks are representative of the time spent within the area being assessed. Home range information is provided in Table 2-17.

Overall, the Tier 2 risk characterization is intended to provide information on the relative risk from different areas within the mine (and downgradient riparian areas) and the relative effects of reclamation type on ECOPC exposures.

HQ calculations were completed both on a panel-by-panel basis and by reclamation type. For riparian receptors, risks were characterized by drainage as represented by the exposure data for seep and riparian locations where samples were collected in drainages adjacent to the mine.

The EPCs used for the Tier 2 HQ calculations are provided in Tables 4-4 and 4-5 for upland receptors and in Appendix B for riparian receptors. Tier 2 exposure estimates are provided in Tables 4-6 through 4-13 for each mine panel and in Tables 4-14 through 4-17 for each reclamation type. Exposure estimates at riparian and seep locations are provided in Appendix C. All HQ calculations are also provided in Appendix C.

When the data are grouped by mine panel, HQs greater than 1.0 were calculated for the following ECOPC/receptor pairs in at least one mine panel and/or in northern Sage Valley. The HQs were calculated using only the lowest LOAEL TRV. This approach was taken because the mine panels are relatively small compared to the Site-wide population areas, and would not be representative of risk to population-level assessment endpoints. HQs were greater than 1.0 for the following ECOPCs:

- Cadmium
    - Deer Mouse – Figure 4-5 (Panel A Area 2, Pole Canyon ODA, Panel D North, Panel E Area 1)
- Copper
    - Coyote – Figure 4-6 (Panel A Area 1, Pole Canyon ODA, Panel D North, Panel D South, Panel E Area 1, Panel E Area 2, Northern Sage Valley)
    - Northern Harrier – Figure 4-7 (Panel A Area 1, Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South, Panel E Area 1, Panel E Area 2, Northern Sage Valley)
- Lead
    - Northern Harrier – Figure 4-8 (Panel A Area 1, Pole Canyon ODA, Panel D North, Panel D South, Panel E Area 1, Panel E Area 2)

- Selenium
    - American Robin – Figure 4-9 (Panel A Area 1, Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South, Panel E Area 1, Panel E Area 2, Northern Sage Valley)
    - Coyote – Figure 4-10 (Panel A Area 1, Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South)
    - Deer Mouse – Figure 4-11 (Panel A Area 1, Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South, Panel E Area 1, Panel E Area 2, Northern Sage Valley)
    - Eastern Cottontail – Figure 4-12 (Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South)
    - Mule Deer – Figure 4-13 (Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South)
    - Northern Bobwhite – Figure 4-14 (Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South)
    - Northern Harrier – Figure 4-15 (Panel A Area 1, Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South, Panel E Area 1)
- Vanadium
    - American Robin – Figure 4-16 (Panel A Area 1, Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South, Panel E Area 1)
    - Northern Bobwhite – Figure 4-17 (Panel A Area 1, Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South)
    - Northern Harrier – Figure 4-18 (Panel A Area 2, Panel D South)
- Zinc
    - Northern Harrier – Figure 4-19 (Panel A Area 2, Pole Canyon ODA, Panel D North, Panel D South, Northern Sage Valley).

When the data were grouped by reclamation type, HQs greater than 1.0 were calculated for the following ECOPC/receptor pairs in at least one reclamation type. The HQs were calculated using the lowest LOAEL TRV and were greater than 1.0 for the following ECOPCs:

- Cadmium
    - Deer Mouse – Figure 4-20 (No Cover and Topsoil Only)
- Copper
    - Coyote – Figure 4-21 (All Reclamation Types)
    - Northern Harrier – Figure 4-22 (All Reclamation Types)
- Lead
    - Coyote – Figure 4-23 (No Cover)
    - Northern Harrier – Figure 4-24 (All Reclamation Types)
- Selenium
    - American Robin – Figure 4-25 (All Reclamation Types)
    - Coyote – Figure 4-26 (No Cover, Topsoil Over Chert, and Chert Only)
    - Deer Mouse – Figure 4-27 (All Reclmation Types)
    - Eastern Cottontail – Figure 4-28 (No Cover, Topsoil Over Chert, and Chert Only)
    - Mule Deer – Figure 4-29 (No Cover, Topsoil Over Chert, and Chert Only)
    - Northern Bobwhite – Figure 4-30 (No Cover, Topsoil Over Chert, and Chert Only)
    - Northern Harrier – Figure 4-31 (No Cover, Topsoil Over Chert, and Chert Only)
- Vanadium
    - American Robin – Figure 4-32 (No Cover, Topsoil Over Chert, and Chert Only)
    - Northern Bobwhite – Figure 4-33 (No Cover, Topsoil Over Chert, and Chert Only)
    - Northern Harrier – Figure 4-34 (No Cover)
- Zinc
    - Northern Harrier – Figure 4-35 (No Cover and Topsoil Only)

Calculations for seep and riparian locations were conducted on a location-by-location basis because the low number of locations did not allow calculation of meaningful UCLs. In addition, the locations represent unique conditions in that they are either adjacent to seeps within the mine itself or at locations adjacent to the mine within the drainages thus representing unique exposures. The location-specific HQs calculated are not representative of Site-wide or habitat risks. The HQs were calculated to indicate the exposures relative to other parts of the Site.

Both the upland and riparian receptors were assumed to utilize the seep and/or riparian habitats. HQs greater than 1.0 using the lowest LOAEL TRV were calculated for the following ECOPCs at one or more riparian and/or seep sampling location:

- Cadmium – Deer Mouse;
- Chromium – Song Sparrow;
- Copper – Coyote and Northern Harrier;
- Lead – Northern Harrier;
- Manganese – Meadow Vole;
- Molybdenum – Meadow Vole;
- Selenium – All receptors;
- Vanadium – All avian receptors; and
- Zinc – Northern Harrier.

The HQs calculated for riparian and seep locations are summarized in Table 4-22.

For the plant and invertebrate receptors, Tier 2 EPCs were compared to screening benchmarks for all panels (Table 4-23) and reclamation types (Table 4-24). For the ECOPCs with EPCs greater than either benchmark, the potential for effects cannot be ruled out with the screening-level data available for assessment. If additional assessment for potential effects is required in order to make risk management decisions, then additional data may be required at a later date.

The ECOPCs with no HQs greater than 1.0 for any wildlife receptor using the lowest LOAEL TRV in the Tier 2 HQ calculations were not carried forward into the Tier 3 risk characterization and, therefore, risks from those chemicals are considered to be low in all areas of the Site.

For those ECOPCs listed above, risks are discussed in more detail in the Tier 3 risk characterization.

4.3.3 Tier 1 and Tier 2 Risk Characterization Summary

The purpose of the Tier 1 and Tier 2 risk characterization was to further focus the risk characterization discussion on those ECOPC/receptor pairs for which HQs greater than 1.0 using the lowest-LOAEL TRV were calculated. The conclusions reached in Tiers 1 and 2 are summarized in Tables 4-25 and 4-26 for upland receptors and Table 4-27 for riparian receptors. Those ECOPCs with one or more receptors with LOAEL HQs greater than 1.0 were identified as ECOCs and are further discussed in the Tier 3 risk characterization presented in the following sections.

4.3.4 Tier 3 Risk Characterization

The final tier of risk characterization is provided in order to focus the discussion of the potential risk at the Site to those ECOC/receptor pairs for which risk cannot be either screened out using conservative measures or for which HQs greater than 1.0 using the lowest LOAEL TRV were calculated either Site-wide or in smaller Tier 2 exposure domains. Based on the Tier 2 Risk Characterization, cadmium, chromium, copper, lead, manganese, molybdenum, selenium, vanadium, and zinc were identified as ECOCs.

The Tier 3 expands the risk characterization for the ECOCs in two ways:

1) Tier 3 includes location-specific HQs to help identify the locations and areas of the Site that contribute most to exposure and risk. Location-specific HQs reflect the ECOC concentration at only one location and, therefore, are not representative of exposure estimates for populations of ecological receptors because they only represent a small area relative to the home ranges of many receptors. Similarly, for receptors with small home ranges location-specific HQs do not represent risks to populations because the feeding range of individual receptors or exposure for subpopulations that are the assessment endpoints for the SSERA are larger than the area sampled using the RI sampling methodology (Table 2-17).

2) By providing HQ calculations for the range of alternative TRVs available for each ECOPC and other ECOPC-specific data. Because there is considerable variability in the TRVs available for each ECOPC (Table 4-18) due to numerous factors (e.g. chemical form, inter-species variability, differences in exposure regime, sensitivity of endpoints tested, etc.), it is important to provide risk managers with relevant data beyond the NOAEL and lowest-available LOAEL TRVs. Selection of the lowest LOAEL TRV as the sole effects-based TRV provides risk managers with a potentially overly-conservative estimate of risk. This is particularly true where population-level assessment endpoints are the focus of the ERA, as is the case at the Smoky Canyon Mine. Using the most sensitive effects-based TRV may be indicative of effects to a small segment of the population of receptors at the Site and may not indicate effects that could be measured at the population level. Using a range of possible effects-based TRVs allows risk managers to better interpret HQs, particularly those marginally greater than 1.0, than would be possible by using only the lowest LOAEL TRV available. As used in the assessment, additional information about the alternative TRVs and its relationship to the NOAEL and lowest LOAEL TRV are provided.

The Tier 3 Risk Characterization is provided to better support risk management decisions for wildlife receptors by providing a less conservative estimate of toxicity for the ECOPCs than is provided by using only the lowest LOAEL TRV available. Tier 3 is also valuable by providing a more detailed spatial assessment of risks to identify areas where risks may be higher or lower than indicated in the Tiers 1 and 2 Risk Characterizations. Tier 3 HQ calculations are provided for those ECOPC/receptor pairs identified as 'Tier 3' as summarized in Tables 4-26 and 4-27. All HQs calculated in Tier 3 are provided in Appendix C.

Cadmium

Based on the results of the Tier 2 risk characterizations, risks to the deer mouse from cadmium exposure are discussed in more detail in this section. In order to further characterize the potential for risk to the deer mouse from cadmium exposure, HQs were calculated on a location-by-location basis at the upland sampling locations and are provided in Appendix C, Table C-6. HQs calculated at individual sampling locations ranged from less than 1.0 to 1.8 (EPL-17) using the lowest LOAEL TRV.

In addition, three alternative TRVs were identified in Table 4-18. The alternative TRVs ranged from 1.86 to 10 mg/kg BW/day and included the EcoSSL geometric means of the NOAEL TRVs for growth and reproduction and for growth, reproduction, and survival. The lowest of the alternative TRVs was greater than two times higher than the lowest LOAEL TRV for the deer mouse.

Review of the data used in the EcoSSL document indicates that the geometric mean NOAEL TRV was lower than all bounded LOAEL TRVs (i.e., LOAEL TRVs with a NOAEL TRV from the same study) for reproduction and survival endpoints. This indicates that no effects to small mammal reproduction or survival would be predicted at the geometric mean NOAEL TRV.

For growth-based endpoints, the geometric mean NOAEL TRV was greater than the LOAEL TRVs provided in 6 of 23 studies with bounded LOAEL TRVs. A total of 22 of the 23 studies provided in the EcoSSL database were conducted on juvenile or gestating animals and 16 of the 17 studies in which the LOAEL TRV was greater than the geometric mean NOAEL TRV were based on juvenile or gestating animals. This indicates that there is some potential for effects to juvenile or *in utero* small mammals from exposure to cadmium at the geometric mean NOAEL exposure rate; however, there is considerable uncertainty in the likelihood of effects. In addition, extrapolating individual growth-based effects into effects on small mammal populations also has considerable uncertainty.

No HQ greater than 2.0 was calculated using the lowest LOAEL TRV, no HQs greater than 1.0 were calculated in any panel, reclamation type, or seep/riparian location using the geometric mean NOAEL TRV from EcoSSL.

As a result, risk to the deer mouse receptor from exposure to cadmium cannot be conclusively dismissed in the mine panels or reclamation types or at seep DS-7, where HQs greater than 1.0 were calculated using the lowest LOAEL TRV. However, the risk is expected to only be relevant to growth-related effects and, based on the available toxicity data, is not expected to have significant effects on deer mouse populations at the Site.

The HQ calculations are summarized in Table 4-28. Calculations for the upland and riparian sampling locations are provided in Appendix C, Table C-7 and the calculations by panel and reclamation type are provided in Appendix C, Table C-8.

Chromium

Based on the results of the Tier 1 and Tier 2 risk characterizations, additional assessment of chromium risks in Tier 3 was not required for the upland receptors. In the seep and riparian location calculations, Tier 3 risks are further discussed at seep DS-7. At that location, NOAEL HQs were greater than 1.0 for both the American robin and song sparrow receptor. The lowest LOAEL HQ for the American robin was less than 1.0 resulting in a conclusion of low risk at seep DS-7 while the HQ for the song sparrow was slightly higher than 1.0 (HQ = 1.2).

One additional LOAEL TRV (5 mg/kg BW/day) for birds provided in Table 4-18, was derived from the Sample et al. (1996) TRV database, and is based on the same reproduction in black duck study as reported in the EcoSSL database. Both TRVs were calculated from unpublished data collected by Haseltine et al. (unpublished, cited in EPA 2008) and both TRV sources also selected the diet containing 10 mg/kg chromium in food as a NOAEL TRV and the diet containing 50 mg/kg chromium as a LOAEL TRV. The difference in reported LOAEL TRVs between the two data sources was based on the assumed ingestion rate of a black duck since the study did not report actual food ingestion. The EcoSSL database assumed a food ingestion rate equal to 61.9 g/day while Sample et al. (1996) assumed 125 g/day food ingestion based on data from a Heinz et al. (1989) study of black ducks. The body weight used for these TRV sources was also slightly different (EcoSSL = 1.17 kg; Sample et al. = 1.25 kg).

Avian toxicity data are limited for chromium and only one additional study containing a bounded LOAEL was identified in the EcoSSL database (LOAEL = 75.4 mg/kg BW/day).

Exposure for the song sparrow at DS-7 was estimated at 3.4 mg/kg BW/day resulting in an HQ equal to 1.2 using the EcoSSL exposure parameters, and an HQ equal to 0.6 using the Sample et al. (1996) exposure parameters (Table 4-29). All HQ calculations for chromium using the alternative TRV are provided in Appendix C, Table C-7.

As a result, while risk to the song sparrow receptor from exposure to chromium cannot be conclusively dismissed at seep DS-7 those risks appear to be limited to small areas. Because the HQs calculated using the low HQs regardless of exposure parameters used to calculate the TRV, significant population-level effects are unlikely.

Copper

Risks to most wildlife receptors from exposure to copper were determined to be low Site-wide, by mine panel and reclamation type, and in northern Sage Valley. However, risks to predators (coyotes and northern harrier) were elevated due to high copper concentrations detected in a number of small mammal samples which are an important dietary component.

Tier 1 HQs greater than 1.0 using the lowest LOAEL TRV were calculated for both the coyote and northern harrier. Since both of these receptors are wide-ranging, the Tier 1 Site-wide risk calculations represent an appropriate scale for overall assessment and risk management decisions for these wide-ranging receptors. The Tier 2 risk characterization identified risks that were low for all receptors except the coyote and northern harrier.

In the seep and riparian sampling locations, risks were similarly low for all receptors except the coyote and northern harrier receptors.

Because of the uncertainty in the copper data in small mammals, Tier 3 risk using alternative TRVs was not conducted for copper.

Coyote

As a first step in the Tier 3 risk characterization, HQs based on the lowest LOAEL TRV were calculated for each sampling location (Figure 4-36). HQs greater than 1.0 were calculated for Panel A (n = 3), Panel D (n = 7), Panel E (n = 3), Pole Canyon (n = 2), and northern Sage Valley (n = 5). All HQ calculations are provided in Appendix C, Table C-6.

The calculated exposure to the coyote receptor is heavily influenced by copper concentrations in small mammal tissues. As the primary prey type ingested, small mammal tissues make up the large majority (90%) of the exposure for the coyote (Table 2-14).

Copper concentrations in a number of small mammal samples collected in 2010 were highly elevated with Site-wide concentrations ranging from 11.9 to 3,900 mg/kg with a mean concentration equal to 178 ± 524 mg/kg (mean ± standard deviation). The highest HQs shown in Figure 4-36 are directly related to the small mammal samples with the highest concentrations and the data exhibit a large degree of variability.

The copper concentrations were much higher than anticipated and the samples containing higher copper concentrations show no spatial pattern or relationship with any mine panel or reclamation type. As a point of comparison, copper concentrations in small mammals collected from the Site in 2004 (49 samples) as part of the SI (NewFields 2005) and summarized in the RI Report (Formation 2014) were much lower and less variable, ranging from 10.5 to 17 mg/kg with an average concentration equal to 14 ± 1.6 mg/kg.

No apparent field or laboratory errors were identified in the data validation process. However, the extremely high concentrations and differences from SI samples and area-wide samples (TTEMI 2002) suggest that the concentration estimates may be anomalous and, therefore, the currently available exposure estimates may be uncertain. Because of the uncertainties associated with the small mammal copper data, additional sampling of small mammal tissues should be considered as part of the RI/FS process prior to making risk management decisions if those decisions are driven primarily by the elevated copper concentrations in question. Resampling could focus on any of the mine panels or reclamation types, including northern Sage Valley and the riparian sampling locations.

Northern Harrier

Risks to the northern harrier receptor were somewhat higher than those estimated for the coyote due to the harrier's higher percentage of small mammal ingestion (100% vs. 90%).

HQs using the lowest LOAEL TRV were also calculated at each sampling location and the results are provided in Figure 4-37. HQs greater than 1.0 were calculated in Panel A (n = 11), Panel D (n = 8), Panel E (n = 12), Pole Canyon (n = 4), and northern Sage Valley (n =6). The highest HQ (HQ = 60) was calculated at Pole Canyon location PCO-14. All HQ calculations are provided in Appendix C, Table C-6.

As with the coyote, harrier HQs were driven by elevated copper concentrations in small mammal tissues which are uncertain. Therefore, it is recommended that resampling of small mammals for copper be considered prior to making risk management decisions for the harrier if the elevated copper concentrations in question are a primary focus of the decision.

Lead

Risks to wildlife receptors from exposure to lead were determined to be low for most receptors both Site-wide, by mine panel and reclamation type, and in northern Sage Valley.

In the Tier 1 risk characterization, HQs greater than 1.0 using the lowest LOAEL TRV were calculated for the northern harrier receptor only. Since the harrier is a wide-ranging receptor, the Tier 1 Site-wide risk calculations represent an appropriate scale for assessment.

The Tier 2 risk characterization further refined the risk estimates for the upland receptors. Risks for the coyote were, in general, lower than the harrier with NOAEL HQs less than 1.0 in Panel A Area 2, Panel D North, Panel D South, Panel E Area 1, Panel E Area 2, and northern Sage Valley as well as in all reclamation types except the areas with no cover.

Risks for the northern harrier were determined to be low (NOAEL HQs less than 1.0) in Panel A Area 2 and LOAEL HQs were less than 1.0 in northern Sage Valley.

In the seep and riparian sampling locations, NOAEL HQs for the coyote were less than 1.0 at all locations. However risks for the northern harrier were higher with HQs greater than 1.0 based on the lowest LOAEL TRVs for samples from Hoopes Spring channel (HS-3), Lower Sage Creek (LS), and Lower South Fork Sage Creek (LSS).

Soil lead concentrations for the Site are relatively low, ranging from about 10 to 17 mg/kg, and may be representative of natural background concentrations. Estimates of natural background concentrations of lead range from about 15 mg/kg for Montana (Hydrometrics 2013) to 20.6 mg/kg for Oregon (Oregon DEQ 2013) for the northeastern areas of the state. If these examples are representative of the mine area, then lead concentrations at the Site are not elevated.

Coyote

As a first step in the Tier 3 risk characterization, HQs based on the lowest LOAEL TRV were also calculated at each sampling location for the coyote and the results are provided in Figure 4-38. An HQ greater than 1.0 (HQ = 1.5) was calculated at only one sample location (PCO-12) on the Pole Canyon ODA. All HQ calculations are provided in Appendix C, Table C-6.

The lowest LOAEL was obtained from the EcoSSL database; additional LOAEL TRVs were provided in the BPF and are shown in Table 4-18. These TRVs included the geometric mean NOAEL TRV for growth and reproduction effects provided in the EcoSSL database (40.73 mg/kg BW/day). The geometric mean TRV was slightly less than two times higher than the lowest bounded LOAEL TRV from the EcoSSL database.

Review of the data reported in the EcoSSL database indicates that there are 50 TRVs containing bounded NOAEL and LOAEL effects values for mammals based on growth, reproductive, and mortality endpoints. For growth-based endpoints, 5 of 20 LOAEL TRVs were less than the geometric mean NOAEL TRV. All but 2 of the TRVs were calculated for sensitive life stage animals (e.g., gestational, nursing, or juveniles).

For reproduction-based endpoints, 5 of 22 LOAEL TRVs were less than the geometric mean NOAEL TRV. For survival-based endpoints, 1 of 8 LOAEL TRVs were less than the geometric mean NOAEL TRV. No bounded LOAEL TRVs were available for canine receptors, and all bounded LOAEL TRVs less than the geometric mean NOAEL TRV were derived from studies of effects on rats.

Based on the available data, the geometric mean NOAEL TRV for mammals from the EcoSSL database may be representative of growth and/or reproductive effects for sensitive individuals in a population. But the available data are heavily weighted toward rodents (rats) which appear to be relatively sensitive to effects from lead exposure. HQs calculated at all sampling locations, mine panels, and reclamation types were less than 1.0 for the coyote using the geometric mean

NOAEL TRV (Table 4-30) and all HQ calculations are provided in Appendix C, Tables C-7 and C-8. None of the other alternative TRVs provided in the BPF and in Table 4-18 were used in the Tier 3 calculations.

While growth, reproduction, or survival effects to coyote populations from lead exposure cannot be ruled out using the currently available data, a more detailed review of the toxicological data for mammals indicates that there is some uncertainty in the applicability of the lowest LOAEL TRV to predict significant effects to canine species.

Northern Harrier

Risks to the northern harrier receptor could not be dismissed either Site-wide (Tier 1) or within most of the mine panels or reclamation types. NOAEL and/or LOAEL HQs in Hoopes Spring channel (HS-3), Lower Sage Creek (LS), and Lower South Fork Sage Creek (LSS) were also greater than 1.0.

For the Tier 3 assessment, HQs were calculated for sampling locations and the results are provided in Figure 4-39. HQs greater than 1.0 were calculated for Panel A (n = 2), Panel D (n = 4), Panel E (n = 4), Pole Canyon ODA (n = 2), and northern Sage Valley (n = 1). The highest HQ (HQ = 13) was calculated for the Pole Canyon ODA at sample location PCO-12. All HQ calculations are provided in Appendix C, Table C-6.

Five additional TRVs were identified in the BPF and ranged from 8.75 to 16.93 mg/kg BW/day and included the EcoSSL geometric means of the NOAEL TRVs for growth and reproduction and for growth, reproduction, and survival.

The geometric mean NOAEL for growth and reproduction (10.94 mg/kg BW/day) was approximately equal to five times the lowest bounded LOAEL TRV identified in the EcoSSL database.

Review of the data reported in the EcoSSL database indicates that there are 18 TRVs containing bounded NOAEL and LOAEL effects values for birds based on growth, reproductive, and mortality endpoints. For growth-based endpoints, 0 of 10 LOAEL TRVs were less than the geometric mean NOAEL TRV and all of the available TRVs were calculated using juvenile animals. For reproduction-based endpoints, 3 of 5 LOAEL TRVs were less than the geometric mean NOAEL TRV and for survival-based endpoints, 0 of 3 LOAEL TRVs were less than the geometric mean NOAEL TRV.

One NOAEL TRV study was available for raptor reproduction (American kestrel), with no effects observed at the highest dose level (NOAEL = 12.0 mg/kg BW/day). For growth effects, one study showed effects to American kestrel chicks with a NOAEL TRV equal to 25 mg/kg BW/day and a LOAEL TRV equal to 125 mg/kg BW/DAY. A paired NOAEL and LOAEL was also available from the same study for survival effects with a NOAEL equal to 125 mg/kg BW/day and a LOAEL equal to 625 mg/kg BW/day (Hoffman et al. 1985).

A NOAEL for American kestrel growth and survival in adults was also provided and was equal to 54.3 mg/kg BW/day and no effects were observed at the highest dose level (Custer et al. 1984).

HQs calculated using the geometric mean NOAEL TRV are provided in Table 4-30 and in Appendix C, Tables C-7 and C-8. None of the other alternative TRVs provided in the BPF and in Table 4-18 were used in the Tier 3 calculations.

Predicted exposure on a Site-wide basis was lower than the geometric mean NOAEL TRVs resulting in HQs less than 1.0. In Panel A Area 1 and the Pole Canyon ODA, exposure was predicted to be higher than the geometric mean TRV. Predicted exposure in Panel E Area 1 was greater than the geometric mean NOAEL TRV for growth and reproduction but lower for the geometric mean NOAEL TRV for growth, reproduction, and survival (16.93 mg/kg BW/day).

In the riparian areas, predicted exposure at location LS in Lower Sage Creek was higher than the geometric mean NOAEL for growth and reproduction but not the geometric mean for growth, reproduction and survival.

All exposures calculated were less than the lowest LOAEL provided for American kestrels in the EcoSSL database (125 mg/kg BW/day).

Lead exposure to the northern harrier is driven by the small mammal concentration. The average lead concentrations in the small mammals collected at PCO-12 (253 mg/kg), PCO-14 (104 mg/kg), APL-21 (153 mg/kg), EPL-12 (115 mg/kg), and LS (112 mg/kg) were considerably higher than those detected at other sampling locations and heavily influenced the exposure calculations in their respective mine panels and reclamation types.

On a Site-wide basis, while risk to the northern harrier receptor cannot be conclusively ruled out based on exposures that exceeded the lowest LOAEL TRV, predicted exposures were less than the alternative TRVs as discussed above. Within the Pole Canyon ODA, Panel A Area 1, and Panel E Area 1 areas, and at LS, risks to the northern harrier may be somewhat higher than elsewhere but are driven by a few small mammal samples with elevated lead concentrations.

Given the foraging range of northern harrier, it is unlikely that more than single birds would feed exclusively within any of the mine panels or drainages at the Site. As a result, the Site-wide exposure scenario is likely to be the best predictor of risk to the local subpopulation of northern harrier receptor.

Manganese

Based on the results of the Tier 1 and Tier 2 risk characterizations, in the seep and riparian location calculations, at HS-3 and LSS, LOAEL HQs calculated using the lowest LOAEL TRV for the meadow vole receptor were slightly greater than 1.0. The HQ at HS-3 was equal to 1.5 and the HQ at LSS was equal to 1.4. The LOAEL HQs at all other locations were less than 1.0.

No additional TRVs were available for manganese, so no additional assessment was conducted for Tier 3. As a result, while risk to the meadow vole receptor from exposure to manganese cannot be conclusively dismissed at riparian locations HS-3 and LSS, those risks appear to be limited to small areas and exposure is predicted to be only slightly higher than the lowest LOAEL TRV indicating that significant population-level effects are unlikely.

Molybdenum

Based on the results of the Tier 1 and Tier 2 risk characterizations, for the seep and riparian location calculations, at location LP-PD, the LOAEL HQ was equal to 1.4 using the lowest LOAEL TRV. The LOAEL HQs at all other locations were less than 1.0.

No additional TRVs were available for molybdenum, so no additional assessment was conducted for Tier 3. As a result, while risk to the meadow vole receptor from exposure to molybdenum cannot be conclusively dismissed at riparian location LP-PD those risks appear to be limited to small areas and exposure is predicted to be only slightly higher than the lowest LOAEL TRV indicating that significant population-level effects are unlikely.

Selenium

Selenium was identified as an ECOPC for all receptors considered in the SSERA.

In the Tier 1 risk characterization, HQs greater than 1.0 using the lowest LOAEL TRV were calculated for all upland receptors on a Site-wide basis.

The Tier 2 risk characterization similarly resulted in HQs greater than 1.0 for all upland receptors in all mine panels. Risks were lower to all receptors within the areas reclaimed using the Dinwoody cover than for other reclamation types. In the seep and riparian sampling locations, HQs greater than 1.0 for at least one receptor were calculated at all sampling locations, with the highest risks calculated at seeps DS-7 and ES-4.

Location-by-Location Calculations

As the first step in the Tier 3 risk characterization, location-by-location HQs were calculated based on the lowest LOAEL TRVs for each receptor in Figures 4-40 through 4-46 using the lowest LOAEL TRVs.

The spatial patterns shown in the HQs in these figures closely match the HQs presented in the Tier 2 assessment.

The American robin, deer mouse, and northern harrier receptors are highlighted because they are modelled to have the highest exposure relative to their TRVs in each of the major feeding guilds (avian and mammalian omnivores and carnivores).

Alternative TRVs

HQs also were calculated using the geometric mean NOAEL TRVs for the American robin, deer mouse, and northern harrier receptors. The HQs for the receptors are shown by mine panel (Figures 4-47 through 4-49) and reclamation type (Figures 4-50 through 4-52). All HQ calculations are provided in Appendix C, Table C-8.

For selenium, the NOAEL TRV and the lowest LOAEL TRVs obtained from the EcoSSL database are very similar and do not provide risk managers with a good estimate of the potential range of exposure at which effects may be observed. This is especially evident for mammalian receptors that have nearly identical NOAEL (0.143 mg/kg BW/day) and lowest LOAEL (0.145 mg/kg BW/day) TRVs. While this may be indicative of a very robust toxicological dataset, nearly identical NOAEL and LOAEL TRVs provide risk managers with only a limited view of the potential for risk.

In order to provide more useful information on the range of potential risks, the Tier 3 assessment also utilized the geometric mean NOAEL TRV presented in the EcoSSL document as a comparison point. These values provide an estimate of the mean exposure rate across all of the sublethal growth and reproduction endpoints in the database. This TRV may provide risk managers with a better estimate of the average exposure rate across species that have been shown to have no effects. Because the TRV is higher than the lowest LOAEL, some level of effects are possible at or below the TRV, however, those effects should be more closely considered.

For mammals, the EcoSSL database provides 8 reproduction-based NOAEL and LOAEL TRV pairs, 44 growth TRV pairs, and 26 survival TRV pairs. The geometric mean NOAEL for reproduction and growth (0.437 mg/kg BW/day), and reproduction, growth, and survival (0.543 mg/kg BW/day) were three to four times higher than the lowest bounded LOAEL available in the

EcoSSL document. None of the other alternative TRVs provided in the BPF and in Table 4-18 were used in the Tier 3 calculations.

Of the available studies, 1 of 8 reproductive LOAELs and 4 of 26 survival LOAELs were lower than one or both of the geometric mean NOAELs. For the growth-related endpoints, between 25 and 33 percent of the LOAELs were less than the geometric mean NOAELs. All of the growth-related LOAEL TRVs were for juvenile animals and were primarily based on effects to rats and pigs.

Exceedance of the geometric mean NOAEL TRVs may represent exposure levels related to growth, reproduction, and survival effects for sensitive individuals in a population. Based on the data from the EcoSSL compilation, the geometric mean NOAEL may be an appropriate TRV for population-level assessment endpoints, instead of the lowest bounded LOAEL. The geometric mean NOAEL is still very conservative, and other factors including habitat and the extent of local populations are much bigger factors in determining whether ecologically meaningful adverse impacts on a population are expected.

For birds, the EcoSSL database provides 8 reproduction-based NOAEL and LOAEL TRV pairs, 16 growth TRV pairs, and 19 survival TRV pairs. The geometric mean NOAEL for reproduction and growth (0.606 mg/kg BW/day) and reproduction, growth, and survival (0.85 mg/kg BW/day) were slightly higher than the lowest bounded LOAEL available in the EcoSSL document. None of the other alternative TRVs provided in the BPF and in Table 4-18 were used in the Tier 3 calculations.

Of the available studies, 6 of 8 reproductive LOAELs and 2 of 19 survival LOAELs were lower than one or both of the geometric mean NOAELs. For the growth-related endpoints, only 2 of 16 of the LOAELs were less than the geometric mean NOAELs. As a result, the geometric mean NOAEL TRVs may be the most appropriate TRV for population-level assessment endpoints. However, because of the similarity of the lowest LOAEL TRV (0.368 mg/kg BW/day) and the geometric mean for reproduction and growth (0.606 mg/kg BW/day), it would be difficult to distinguish different levels of risk between the two TRVs.

In general, for most receptors, exposure is highest in Panel A Area 2, Panel D North and South, and the Pole Canyon ODA (Table 4-31) which represent areas where exposure to selenium-bearing overburden materials is expected to be highest.

HQs are considerably lower in northern Sage Valley, Panel E Areas 1 and 2, and Panel A Area 1 than in Panel D North and South and the Pole Canyon ODA. The calculated exposure and HQs in Panel A Area 1 and in Panel E Areas 1 and 2 were very similar to those calculated in northern Sage Valley. Since the potential for mining-related impacts throughout the majority of northern Sage Valley is expected to be low, these HQs are expected to be related to baseline conditions within the area and are less likely to be indicative of mining-related exposure.

Similarly, when HQs are calculated by reclamation type, they are considerably lower in the Dinwoody reclamation area and highest in the areas with no cover and topsoil only, with the topsoil over chert reclamation area intermediate between the two. Panel E Area 2 was reclaimed using a Dinwoody cover which included a layer of chert as a capillary break. Concentrations in all media within the Dinwoody reclaimed area (Panel E Area 2) were similar to those measured outside of the mine-disturbed area in northern Sage Valley indicating that they are similar to exposures predicted outside of the mining area. As a result, HQs calculated for areas outside of northern Sage Valley and the Dinwoody cover areas may be representative of areas without mine-related impacts, and good comparison points to estimate the potential for incremental risk from mine impacts.

Selenium concentrations in all media at Panel A Area 1 and Panel E Area 1 were slightly higher than in northern Sage Valley and the Dinwoody reclamation type, but were still considerably lower than in the other areas of the Site. Reclamation efforts in those areas appear to have been successful in significantly reducing exposure. However, since HQs greater than 1.0 were still calculated in those areas, it is possible that the exposure models either over-predict exposure or the TRVs selected over-predict the potential for risk to selenium at low concentrations.

HQs calculated in Panel A Area 2, Panel D North and South, and the Pole Canyon ODA are considerably higher than those calculated in northern Sage Valley or in the areas with more extensive reclamation. Risks to receptors inhabiting these areas are, therefore, predicted to be higher.

Background selenium concentrations estimated for other sites in the region were determined to be approximately equal to 2 mg/kg in upland and riparian soils at another local mine site (MWH, 2013) and on a broader spatial scale from 1.2 to 7 mg/kg at some other mine sites within the region (Arcadis and AECOM 2014). It is unknown to what extent the incremental risk from the Site is increased over what would be expected to occur if the Site were not present. However, given the very low soil selenium concentrations observed in the Dinwoody reclamation type (95UCL = 0.5 mg/kg) and in Panel A Area 1 (95UCL = 3.3 mg/kg), Panel E Area 1 (95UCL = 3.1 mg/kg), Panel E Area 2 (95UCL = 0.5 mg/kg), and northern Sage Valley (95UCL = 0.62 mg/kg), it is likely that the soil concentrations within those areas are similar to what would be expected in background soil.

As a result, the potential for population-level effects appears to be low within northern Sage Valley, Panel A Area 1, and Panel E Areas 1 and 2. The potential for effects within the Panel A Area 2, Pole Canyon ODA, and Panel D North and South areas is higher than in the other areas of the Site.

Similarly, in the areas reclaimed using the Dinwoody reclamation type (i.e., Panel E Area 1), the potential for population-level effects appears to be low. The potential for effects is slightly higher in the topsoil over chert reclamation type and highest in the no cover and topsoil only reclamation types.

HQs for the riparian receptors at all locations using the geometric mean NOAEL TRV are provided in Table 4-32. HQs greater than 1.0 were calculated for most of the receptors at seeps DS-7 and ES-4 as well as at riparian location LP-PD adjacent to the Pole Canyon ODA. Using the alternative TRVs, no HQs greater than 1.0 were calculated at seep ES-3 or at riparian locations LS and LSm. HQs greater than 1.0 were calculated for several receptors at each of the remaining riparian sampling locations but were considerably lower than calculated at LP-PD.

Risks to riparian receptors were highest at seep locations DS-7 and ES-4 as well as at riparian location LP-PD.

Risks to riparian receptors cannot be conclusively dismissed at any seep or riparian sampling location. However, the potential for population-level risks at seep ES-3 and riparian locations LS and LSm appears to be low. The potential for effects at Hoopes Spring and LSS appears to be slightly higher, but still relatively low. The potential for risks at LP-PD and seeps DS-7 and ES-4 is the highest observed for locations sampled under the RI.

Vanadium

In the Tier 1 risk characterization, HQs greater than 1.0 using the lowest LOAEL TRV were calculated for the American robin receptor only. The Tier 2 risk characterization further refined the risk estimates for the upland receptors. The NOAEL HQs calculated for the American robin receptor were less than 1.0 in northern Sage Valley and in the Dinwoody reclamation type, but not elsewhere on the Site or in any of the other reclamation types.

Tier 2 NOAEL HQs for the northern bobwhite were less than 1.0 in northern Sage Valley and in Panel E Areas 1 and 2, but not in Panels A, D, or the Pole Canyon ODA. Correspondingly, NOAEL HQs were less than 1.0 in the Dinwoody reclamation type only.

Finally, for the northern harrier, NOAEL HQs less than 1.0 were calculated in Panel A Area 1, Panel D North, both areas of Panel E, Pole Canyon ODA, and northern Sage Valley. The lowest LOAEL HQ was greater than 1.0 in Panel A Area 2 and Panel D South.

NOAEL HQs greater than 1.0 but lowest LOAEL HQs less than 1.0 were calculated in the topsoil only reclamation type and NOAEL HQs were less than 1.0 in the topsoil over chert and Dinwoody reclamation types. The HQs calculated using the lowest LOAEL TRV were greater than 1.0 in the reclamation areas with no cover.

In the seep and riparian sampling locations, NOAEL and LOAEL HQs were less than 1.0 at all locations except:

- American Robin – seeps DS-7 and ES-3;
- Mallard – LP-PD, LS, LSm, LSS;
- Red-Winged Blackbird – seeps DS-7 and ES-3; and
- Song Sparrow – seeps DS-7 and ES-3 and LSS.

American Robin

HQs were calculated at each of the sampling locations and the results are shown in Figure 4-53. HQs greater than 1.0 were calculated in Panel A (n = 9), Panel D (n = 10), Panel E (n = 3), and Pole Canyon (n = 3). No HQs greater than 1.0 were calculated in northern Sage Valley.

Three additional TRVs were identified (Table 4-18) ranging from 1.19 to 11.4 mg/kg BW/day. All of the additional TRVs were NOAEL values and included the EcoSSL geometric means of the NOAEL TRVs for growth and reproduction (1.19 mg/kg BW/day) and for growth, reproduction, and survival (1.9 mg/kg BW/day).

For birds, the EcoSSL database provides 6 reproduction-based NOAEL and LOAEL TRV pairs, 14 growth TRV pairs, and 8 survival TRV pairs. The geometric mean NOAEL for reproduction and growth (1.19 mg/kg BW/day) and reproduction, growth, and survival (1.9 mg/kg BW/day) were three to four times higher than the lowest bounded LOAEL available in the EcoSSL document.

Of the available studies, 2 of 6 reproductive LOAELs and 0 of 8 survival LOAELs were lower than one or both of the geometric mean NOAELs. For the growth-related endpoints, 3 of 14 LOAELs were less than the geometric mean NOAEL for growth and reproduction. Eight of the growth-related LOAEL TRVs that were higher than the geometric mean NOAEL were for juvenile animals and were all based on effects to chickens.

One growth study was available for a non-chicken endpoint (Japanese quail) and showed no effects to growth in juveniles (46.1 mg/kg BW/day), reproduction (39.0 mg/kg BW/day), or survival (98.7 mg/kg BW/day (Hafez and Kratzer [1976]). Several studies of vanadium effects on duck mortality were also available with NOAEL TRVs equal to 12.0 and 13.4 mg/kg BW/day.

Based on the above data from the EcoSSL compilation, the geometric mean NOAEL may be an appropriate TRV for population-level assessment endpoints, instead of the lowest bounded LOAEL. The geometric mean NOAEL is still very conservative, and other factors including habitat and the extent of local populations are much bigger factors in determining whether ecologically meaningful adverse impacts on a population are expected.

Using the geometric mean NOAEL TRV for growth and reproduction (1.19 mg/kg BW/day), HQs equal to 1.0 were calculated in Panel A Area 2 and Panel D North and South; all other HQs were less than 1.0. All HQ calculations are provided in Appendix C, Tables C-7 and C-8. None of the other alternative TRVs provided in the BPF and in Table 4-18 were used in the Tier 3 calculations.

When HQs were calculated by reclamation type, an HQ equal to 1.3 was calculated in the no cover reclamation type using the geometric mean NOAEL for growth and reproduction. No other HQs greater than 1.0 were calculated in any reclamation type using the geometric mean NOAEL TRV

At the seep and riparian locations, all calculated HQs were less than 1.0 except at seep DS-7, where the HQ calculated using the geometric mean NOAEL for growth and reproduction was equal to 1.1.

Risk to the American robin receptor from exposure to vanadium cannot be conclusively dismissed in several areas of the Site. However, based on the low HQs calculated and the lack of significant exceedances of the geometric mean NOAEL TRVs, significant population-level effects are unlikely.

Northern Bobwhite

HQs were calculated for the northern bobwhite at each of the sampling locations and the results are provided in Figure 4-54. HQs greater than 1.0 were calculated in Panel A (n = 6), Panel D (n = 6), Panel E (n = 1), and the Pole Canyon ODA (n = 2). No HQs greater than 1.0 were calculated in northern Sage Valley.

As discussed for the American robin, HQs were also calculated using the geometric mean NOAEL TRV from the EcoSSL database. Using the geometric mean NOAEL TRV for growth and reproduction (1.19 mg/kg BW/day), HQs were less than 1.0 in all mine panels and reclamation types (Table 4-33). At the seep and riparian locations, all calculated HQs were less than 1.0 using the geometric mean NOAEL for growth and reproduction. All HQ calculations are provided in Appendix C, Tables C-7 and C-8.

Risk to the northern bobwhite receptor from exposure to vanadium cannot be conclusively dismissed in several areas of the Site. However, based on the low HQs calculated and the lack of significant exceedances of the geometric mean NOAEL TRVs, significant population-level effects are unlikely.

Northern Harrier

HQs were calculated for the northern harrier at each of the sampling locations and the results are provided in Figure 4-55. HQs greater than 1.0 were calculated in Panel A only (n = 4). No HQs greater than 1.0 were calculated at any other upland sampling location.

As discussed for the American robin, HQs were also calculated using the geometric mean NOAEL TRV from the EcoSSL database. Using the geometric mean NOAEL TRV for growth and reproduction (1.19 mg/kg BW/day), HQs were less than 1.0 in all mine panels and reclamation types (Table 4-33). All HQ calculations are provided in Appendix C, Tables C-7 and C-8.

At the seep and riparian locations, all calculated HQs were also less than 1.0 using the geometric mean NOAEL for growth and reproduction.

Risk to the northern harrier receptor from exposure to vanadium cannot be conclusively dismissed within Panel A. However, based on the low HQs calculated and the lack of significant exceedances of the geometric mean NOAEL TRVs, significant population-level effects are unlikely.

Mallard

HQs greater than 1.0 using the lowest LOAEL were calculated for the mallard receptor at riparian sampling locations LP-PD, LS, LSm, and LSS. The HQs were recalculated using the range of alternative TRVs discussed in the previous sections and are provided in Table 4-18.

Using the geometric mean NOAEL for growth and reproduction resulted in an HQ equal to 1.0 at LS and less than 1.0 at all other sampling locations. All HQs were less than 1.0 using the other alternative TRVs (Table 4-33). All HQ calculations are provided in Appendix C, Table C-8.

Risk to the mallard receptor from exposure to vanadium cannot be conclusively dismissed at several of the riparian sampling locations. However, based on the low HQs calculated and the lack of significant exceedances of the geometric mean NOAEL TRVs, significant population-level effects are unlikely.

Red-Winged Blackbird

HQs greater than 1.0 using the lowest LOAEL were calculated for the red-winged blackbird receptor at seep sampling locations DS-7 and ES-3. The HQs were recalculated using the range of alternative TRVs discussed in the previous sections and are provided in Table 4-18.

Using the geometric mean NOAEL TRV resulted in HQs less than 1.0 at both locations (Table 4-33). Significant population-level effects are unlikely.

Song Sparrow

HQs greater than 1.0 using the lowest LOAEL were calculated for the song sparrow receptor at seep sampling locations DS-7 and ES-3 and riparian sampling location LSS. The HQs were recalculated using the range of alternative TRVs discussed in the previous sections and are provided in Table 4-18.

Using the geometric mean NOAEL TRV resulted in HQs less than 1.0 at seep location ES-4 and riparian location LSS. At seep location DS-7, an HQ equal to 1.1 was calculated using the geometric mean NOAEL TRV for growth and reproduction (Table 4-33).

Risk to the song sparrow receptor from exposure to vanadium cannot be conclusively dismissed at seeps DS-7 and ES-4, as well as at riparian sampling location LSS. However, based on the low HQs calculated and the lack of significant exceedances of the geometric mean NOAEL TRVs, significant population-level effects are unlikely.

Zinc

In the Tier 1 risk characterization, the Site-wide HQ for the northern harrier was equal to 1.2 using the lowest LOAEL. The Tier 2 risk characterization NOAEL HQs for the northern harrier were less than 1.0 in the Dinwoody and topsoil over chert reclamation types as well as in Panel A Area 1 and Panel E Areas 1 and 2. HQs greater than 1.0 were calculated in Panel A Area 2, Panel D North, Panel D South, Pole Canyon ODA, and northern Sage Valley, as well as in the no cover and topsoil only reclamation types.

In the seep and riparian location calculations, NOAEL HQs were less than 1.0 at all locations for the meadow vole. For the northern harrier receptor, HQs greater than 1.0 were calculated using the lowest LOAEL TRV at riparian locations LSm and LSS and at seeps DS-7, ES-3, and ES-4.

HQs were calculated for the northern harrier at each of the sampling locations and the results are provided in Figure 4-56.

The range of TRVs identified in the BPF is shown in Table 4-18. Based on discussions with the Agencies as part of the SSERA completion process, it was determined that the lowest LOAEL TRV from EcoSSL was incorrectly calculated and that the correct lowest LOAEL TRV was equal to 79.8 mg/kg BW/day (Gibson et al. 1986). The error in calculation of the EcoSSL lowest LOAEL TRV was due to calculations that failed to include the zinc concentration in the food prior to the addition of zinc acetate as part of the study. After correction, the revised lowest LOAEL TRV (Appendix C, Table C-9) was used to characterize risks to the northern harrier.

Using the revised lowest-LOAEL TRV, HQs at seep and riparian locations were less than 1.0 at ES-4, HS-3, HS-CC1, HS-CC2, LP-PD, and LS. HQs greater than 1 were calculated at locations DS-7 (HQ = 1.3), ES-3 (HQ = 1.1), LSm (HQ = 1.4), and LSS (HQ = 2.0). Using the same TRV, HQs were greater than 1.0 but less than 2.0 in Panel A Area 2 (HQ = 1.3), Panel D South (HQ = 1.4), North Sage Valley (HQ = 1.8), and Pole Canyon (HQ = 1.8) (Table 4-34). All HQ calculations are provided in Appendix C, Tables C-7 and C-8. None of the other alternative TRVs provided in the BPF and in Table 4-18 were used in the Tier 3 calculations.

Risk to the northern harrier receptor from exposure to zinc cannot be conclusively dismissed at the seeps and riparian areas with HQs greater than 1.0 using the revised lowest LOAEL TRV. Similarly, the potential for risk cannot be conclusively dismissed in the mine panels or reclamation types with HQs greater than 1.0. However, given the small exceedances of the revised lowest LOAEL and the fact that the highest HQ calculated at the Site was located in North Sage Valley where no known sources of mine-related ECOCs are present, significant population level effects to the northern harrier receptor are unlikely.

4.3.5 Additional Risk Characterization

The expanse of this Site yields some unique characteristics which, combined with active mining and multiple sampling programs being conducted both before and after the RI data collection effort, may lead to some unique situations where risk characterization is not as clearly defined and pending uncertainty may be present. These unique situations are discussed below.

Risk to Burrowing Receptors

The BPF indicates that the SSERA should also consider the potential overestimation or underestimation of risks to burrowing animals from incidental exposure to subsurface soils within the root zone. No subsurface soil data were collected for the RI; however, a limited set of subsurface soil data (6 to12 inches) were collected as part of the 2004 SI (NewFields 2005) and were discussed in the RI (Formation 2014).

Subsurface soil data were collected in six of the exposure domains (Figure 4-57). Data are available in Panel A Area 2 (n = 5), Panel D North (n = 10), Panel D South (n = 9), Pole Canyon ODA (n = 9), and northern Sage Valley (n = 20). No subsurface soil data were available specifically for Panel A Area 1, Panel B, Panel C, or Panel E.

In general, where data were available, mean subsurface soil concentrations were similar to mean surface soil concentrations. In Panel A Area 2, average selenium concentrations in subsurface soils were lower (12.5 mg/kg) than in surface soils (30.5 mg/kg). In Panel D, concentrations were similar with average concentrations in Panel D North (8.86 vs 9.0 mg/kg) and Panel D South (15.7 vs 14.2 mg/kg) in subsurface and surface soils respectively. In northern Sage Valley, mean selenium concentrations were low in both subsurface (0.43 mg/kg) and surface soils (0.94 mg/kg).

Subsurface selenium concentrations were only higher than in surface soils on average within the Pole Canyon ODA (34.3 vs. 17.2 mg/kg). However, as indicated in the RI, selenium concentrations in subsurface soils were elevated at two locations (DT-30 and PT-12), relative to the other results for subsurface soils. Similar concentrations were measured in surface (65.9 mg/kg) and subsurface (92.7 mg/kg) soils at DT-30, but concentrations at PT-12 were very different between the surface (29 mg/kg) and subsurface (148 mg/kg).

Based on these results, the SSERA neither overestimates nor underestimates risk for those animals that may be incidentally exposed to subsurface soil with the exception of the locations within the Pole Canyon ODA discussed above.

Risks in Panels B and C

Panels B and C were identified in the BPF (Formation 2013) for consideration in the SSERA, but data were not collected in either mine panel under the RI. As discussed in the RI, mining was recently completed at Panel C and is ongoing at Panel B.

No surface soil samples were collected under either the SI or the RI within Panel B; however, eight samples were collected in 2011 (n=4) and 2012 (n=4) as part of the CEMPP (Formation 2012d) reclamation monitoring program. Selenium concentrations in those samples ranged from 0.66 to 1.95 mg/kg with a mean concentration of 1.3 ± 0.47 mg/kg.

In Panel C, 20 samples were collected in 2011 (n=10) and 2012 (n=10) as part of the CEMPP (Formation 2012d) reclamation monitoring program. Selenium concentrations in those samples ranged from 1.25 to 7.51 mg/kg with a mean concentration of 3.72 ± 1.39 mg/kg.

Panel A Area 1 was reclaimed using the same methods as were prescribed for Panel B and Panel C (BLM and USFS 2002). Therefore, data from Panel A Area 1 can be used to represent potential surface soil selenium concentrations for Panels B and C in the SSERA. As a result, the risk-based conclusions reached for Panel A Area 1, can also be applied to Panels B and C with moderate to low uncertainty (i.e., high confidence) based on the results of the limited data available and the use of the same reclamation techniques in these areas.

Riparian Risk in Pole Canyon Creek

Portions of northern Sage Valley have historically been irrigated with water that may have had elevated selenium concentrations. A number of soil and vegetation samples were collected to provide a general characterization of the nature and extent of selenium concentrations in Sage Valley beyond the RI data discussed in Section 4.4.3. An additional 14 samples were collected in 2012 under the RI. One sample was also collected from location LPSV-08 in August 2011 to represent soils upgradient of areas inundated during the spring 2011 high-flow event. Soil and vegetation data only were available from these locations and are provided in Figure 4-57. Selenium concentrations were generally relatively low, ranging from 0.12 to 10.5 mg/kg with a mean concentration of 0.94 ± 1.87 mg/kg. The maximum detection at SV-13 in 2004 was anomalously high when compared to the rest of the soil samples collected from Sage Valley; the next-highest concentration was 3.1 mg/kg at location SV-16.

In 2011, under RI SAP Addendum 02 (Formation 2011d), an additional 19 samples were collected in order to characterize selenium concentrations in the portions of northern Sage Valley along the Pole Canyon Creek drainage pathway that may have been affected by surface water flow from the Pole Canyon ODA toe seep during the spring 2011 high-flow event. Selenium concentrations in most of these samples were similar to the results from the 2010 RI samples, but four samples had distinctly higher concentrations. Selenium concentrations ranged from 0.12 to 52.6 mg/kg with a mean concentration of 7.49 ± 15.9 mg/kg. The mean concentration in the 2011 samples was heavily influenced by four samples containing selenium at concentrations considerably higher than the majority of the Sage Valley soil samples; LPSV-01 (52.6 mg/kg), LPSV-13 (41.1 mg/kg), LPSV-1A (30.8 mg/kg), and LPSV-11(13.9 mg/kg). Samples LPSV-01, LPSV-1A, and LPSV-11 were collected downgradient from the Pole Canyon ODA toe seep (LP-1) which discharged into lower Pole Canyon Creek during the spring 2011 high-flow period (see Figure 6.6-3 of the Final RI Report [Formation 2014]). Sample LPSV-13 was located near sample SV-13 where the highest selenium concentration in northern Sage Valley was measured in the SI (NewFields 2005). Both of these samples were collected from an occasionally wet area along Pole Canyon Creek several thousand feet downstream of the ODA toe seep.

The area surrounding locations SV-13 and LPSV-13 was further sampled in 2012 under RI SAP Addendum 04 (Formation 2012b). Fourteen soil samples were collected in order to determine the extent of elevated selenium concentrations. As shown in Figure 6.6-3 of the Final RI Report (Formation 2014), elevated selenium concentrations in surface soil appear to be related to the occasionally wet area adjacent to Pole Canyon Creek. Selenium concentrations in the four samples closest to LPSV-13 ranged from 17.2 to 33.1 mg/kg indicating higher selenium concentrations in this area while the rest of the sampling locations had low selenium concentrations (ranging from 0.25 to 2.3 mg/kg), similar to all other areas within northern Sage Valley. The samples collected from the area of elevated selenium concentrations (closest to LPSV-13) had concentrations greater than ecological screening-level benchmarks only.

Based on these data, although all media were not available from these sampling locations, risks throughout much of the area potentially receiving water from Pole Canyon Creek can be classified as low, similar to those areas assessed in Section 4.3.4 with several exceptions. The area immediately adjacent to the Pole Canyon ODA, shown in the red inset of Figure 4-54, and the area surrounding sample LPSV-13, shown in the blue shaded area, have higher risk than predicted for the remainder of northern Sage Valley. Both represent relatively small areas, so population risk is not expected to be elevated over what was predicted in Section 4, but risks may be higher to metapopulations of small home range receptors inhabiting those areas.

Risk to Reptiles

Reptiles were identified in the ECSM as a receptor to be evaluated in the SSERA. However, very little exposure and toxicity data are available for reptiles, thus limiting the ability to provide a quantitative assessment of exposure and risk. As indicated in the BPF, exposure and risks calculated for the bird receptors should be considered as representative and protective of the potential risk to the reptile receptors at the Site.

5.0 UNCERTAINTY ANALYSIS

Uncertainty is inherent in every step of the risk assessment process. The general approach in this SSERA has been to err on the side of conservatism. Therefore, the risks in this SSERA are likely to be overestimated rather than underestimated. However, a complete understanding of the uncertainties associated with the risk estimates is crucial to placing the estimated risks into the proper perspective. The uncertainty analysis identifies sources that are common to every risk assessment along with those that are specific to this SSERA.

Ecological risk assessments require assumptions and extrapolations in each step of the assessment that lead to uncertainty in risk prediction and affect projections of true exposure and risk at any site. Accordingly, the key assumptions and uncertainties discussed in the following sections that have the greatest influence on ecological risk assessments include:

- Sampling uncertainty;
- Assumptions regarding exposure probability;
- Sampling uncertainty (uncertainty about spatial distribution of contamination as a consequence of limitations in sampling a site);
- Uncertainty in the selection of ECOPCs;
- Uncertainty in the natural (including seasonal or annual) variability in the species, populations, communities, and ecosystems in question, as well as uncertainty about individual sensitivity to COPCs;
- Uncertainty in risk characterization using laboratory-based toxicity values and the HQ approach;
- Uncertainty in models and parameters used to estimate risk potentials; and
- Uncertainty in assessing background ECOPC concentrations that may relate to calculated risk potentials (incremental risk).

5.1 Sampling Uncertainty

Sampling was conducted according to agency-approved sampling plans and met objectives developed by Simplot and the regulatory Agencies for the RI and risk assessments. In general, sampling was focused on areas of suspected contamination. As a result, more data are available for areas of higher ECOPC concentrations than areas and habitats of the Site that have lower ECOPC concentrations. Unless receptors or receptor subpopulations are restricted to the areas of contamination, this factor tends to overestimate exposure to ECOPCs.

5.2 Assumptions of Exposure Probability

The three-tiered assessment approach used in the SSERA includes several assumptions about exposure to ECOPCs that are conservative for certain sampled conditions and less conservative for others. Examples of these assumptions follow:

- Ponds and detention basins were not considered to provide viable aquatic habitat, therefore, surface water and sediments from these features were not considered in the SSERA as complete exposure pathways for aquatic resources. These areas were considered as complete exposure pathways for non-piscivorous receptors.
- Hoopes Spring and South Fork Sage Creek Springs were included in the SSERA because these springs contribute flow directly to streams with viable aquatic habitat; therefore, the exposure pathway is complete.
- Seeps were generally not included in the aquatic portion of the SSERA because they do not provide viable aquatic habitat.
- Risks to terrestrial and semi-aquatic receptors were only considered at the riparian sampling locations closest downgradient of the mine. These locations were selected as representative of the riparian areas assumed to have the highest exposure to ECOPCs due to release mechanisms and transport mechanisms from the mine panels and ODAs. This assumption provides some uncertainty for receptors inhabiting riparian areas downstream of the mine.

5.3 Uncertainties Associated with ECOPC Selection

Comparing Site RI COPC concentrations to conservative TRVs in the screening-level approach identified ECOPCs for surface soil, sediment, surface water, terrestrial biota, and aquatic biota. This tiered screening effort minimizes the chance that chemicals that are at potentially ecotoxic concentrations will be omitted as ECOPCs. Therefore, the uncertainty associated with the misidentification of ECOPCs in the SSERA is low, as is the potential for not identifying ECOPCs that may be present at ecologically significant concentrations.

5.4 Receptor-Specific Uncertainties

Species, life histories, and behavioral differences can also affect sensitivities to ECOPCs. Exposure was quantified for several indicator species that are intended to be representative of the various groups of species or feeding guilds potentially inhabiting the mine area. Uncertainties are present in the selection of the receptors that could potentially utilize the Site to some degree. The receptors were selected based on several criteria, including their potential to utilize habitats present within the Site, their potential to contact environmental media with elevated concentrations of ECOPCs within those habitats, their potential sensitivities to ECOPCs, and the amount of life history and behavioral information available. These criteria help to decrease the uncertainty associated with selecting the receptors rather than analyzing all of the dominant species present at the Site. In addition, the selection of species for this SSERA focused on those likely to come in contact with contaminated media and, in terms of bioaccumulative chemicals, those media that would be expected to have accumulated the chemical at the highest concentration. However, some species may preferentially forage on certain plant types and, if these have higher ECOPC concentrations than the mix of plant species used for the EPCs, then exposure could be higher. Additionally, some animals have migration routes that may expose them to smaller areas than assumed in the SSERA.

The best available data for the receptors at the Site was used to quantify exposure using life history and behavioral parameters for each receptor to estimate the amount of contact a receptor may have with contaminated media by various exposure routes. The following parameters were used in the exposure models in the SLERA:

- Ingestion rates of food, soil/sediment, and surface water;
- Body weight;
- Dietary proportions of each prey type;
- Feeding habits; and
- Proportion of food obtained from the Site.

Exposure parameters were obtained from guidance published by USEPA (e.g., *Wildlife Exposure Factors Handbook* [USEPA 1993]) or other state and federal agencies, or from well-established literature. The receptors and corresponding exposure parameters were approved by USFS, IDEQ, and USEPA. The use of certain of these exposure parameters derived from studies conducted in habitats and climates different from the landscape of the Site adds uncertainty to the SSERA because the exposure parameter may not reflect actual Site conditions. For example, ingestion rates cited in the Exposure Factors Handbook typically are based on eating habits of laboratory animals with access to an abundant food supply. It is likely that, in a wild setting such as that present at the Site, the same animals would not have access to such an abundant food supply, resulting in a lower actual ingestion rate than cited in the Exposure Factors Handbook. In this case, use of the published values would tend to bias the SSERA toward an overestimation of risk but underestimation is also possible.

While the exposure models used in the SSERA represent the most up-to-date information available for each receptor, the models are essentially simplistic approximations of the overall feeding behaviors of the receptors based on allometric equations used to estimate food intake and professional judgment used to estimate sediment intake. As a result, there is uncertainty inherent in the use of simplistic models to describe the complex interactions that occur in a natural system. Therefore, there is uncertainty involved with estimating exposure to ECOPCs by using simplistic modeling techniques.

5.5 Risk Characterization Uncertainties

There are several and varying levels of uncertainty associated with the process of risk characterization. This section identifies the major risk characterization components as well as uncertainties that apply, regardless of the habitat component for which the risks are being evaluated.

5.5.1 Exposure Point Concentrations

Uncertainty associated with the EPCs comes from the underlying data used to represent or calculate the EPC, and the method used in the calculation. The data on ECOPC concentrations in abiotic media and biota were collected according to agency-approved sampling plans. Upland soil, animal, and plant samples were collected primarily to characterize the nature and extent of contamination, and were focused on areas of suspected contamination. Samples were not collected to represent an unbiased average concentration across the exposure areas. This approach generally results in an overestimate of EPCs. Aquatic organisms were sampled across diverse conditions and habitats and may be more representative of ECOPC concentrations in the Site and downstream areas.

EPCs were calculated at different scales in each of the three tiers of the risk characterization. Site-wide EPCs were calculated in Tier 1 to represent risks to receptor populations across the Site based on the assessment endpoints for the SSERA. For receptors with large home ranges, the Site-wide assessment represents risks to one to several receptors in the local population during the portion of time that they utilize the Site. This may overestimate risk to the local population of receptors of which a significant portion may not utilize the habitat present at the Site more than occasionally. For receptors with smaller home ranges than the Site, Tier 1 risks may be either overestimated or underestimated for the various subpopulations within the Site. Risks to subpopulations inhabiting areas such as those reclaimed using the Dinwoody cover are likely overestimated using the Site-wide EPCs while those subpopulations in areas reclaimed using only topsoil may be underestimated using the Site-wide EPC.

The Tier 2 Risk Characterization utilized EPCs calculated by mine panel as the exposure units, because key remediation decisions are likely to proceed by mine panel. These exposure units are not representative of risks to the population of receptors at the Site as identified by the assessment endpoint, but are more representative of risks to subpopulations of receptors inhabiting each of the mine panels and northern Sage Valley. The Tier 3 Risk Characterization further refines the exposure calculations by using EPCs from each sampling location. These risks are not representative of population-level risks for any receptor, but are useful for identifying areas within each exposure unit where ECOPC concentrations and subsequent exposure may be higher.

When calculating EPCs from sampling data, ProUCL's advanced algorithms for dealing with non-detected samples were used. While this represents a robust method for left-censored data, any approach dealing with non-detect values is associated with some uncertainty, because chemicals that were not detected at the specified detection limit may be absent from the medium or may be present at any concentration below the detection limit. The uncertainty of the EPC will increase as the number of non-detects in a dataset increases, but this uncertainty may not be important if the non-detect concentration is less than the pertinent screening level.

Uncertainties applicable to all chemical measurement data are related to the measurement of representative field concentrations of a chemical. This, in turn, depends upon several factors, including the media in which the chemical is being measured, the form or phase of the chemical being measured, and the concentrations of other chemicals in the media being measured that affect the measurement of the chemical of interest. Uncertainties due to these factors are addressed to the extent possible through good analytical laboratory procedures, and development and implementation of sampling and analysis plans. Despite the sound application of such procedures and plans, these factors do introduce uncertainty into the estimation of ecological risks.

5.5.2 Bioavailability

The bioavailability of an ECOPC creates uncertainty in the risk characterization process. This uncertainty can affect the potential exposure conditions used to estimate bioavailable forms (such as dissolved metals in solution) as well as the toxicity endpoints used to derive risk assessment benchmarks. Bioavailability and ecotoxicity of chemicals are linked to their concentrations and the forms they take (USEPA 1999).

The toxicity of a contaminant is controlled by the following factors:

- Its environmental concentration;
- Site-specific chemistry (especially through ionic solubility and speciation, if a metal or metalloid);
- The physical matrix in which the contaminant is found; and
- The uptake pathways into a target organism from the physical matrix.

All of these factors help to determine the exposure matrix for organisms in the field. Assessment of ecological risks and the potential adverse effects of a contaminant require an understanding of the exposure matrix that may lead to actual uptake by a receptor species. The overall effect of the uncertainties related to unknown bioavailability may overestimate or underestimate the predicted risks by an unknown degree.

For aquatic receptors, wherever possible field data for tissue concentrations were collected. These direct measures of tissues from the field reduce uncertainty about bioavailability and the need for modeling. The effects data from the brown trout adult reproduction studies (Formation 2011c, 2012c) were derived from wild brown trout that accumulated their entire selenium body burden from the streams where they were collected adjacent to the mine-disturbed areas. These streams represented varying exposure conditions based on previous monitoring conducted across many sites in the area. Eggs and milt were stripped from adults and the fertilized eggs were transported to and reared in an off-site laboratory to evaluate effects. Twenty-six separate egg batches were used to begin the study, although only 24 batches successfully hatched. Because the parent fish were exposed to the Site water quality conditions, the Site food web, and biologically relevant environmental selenium concentrations emanating from Site sources, there is certainty that the bioavailability factors that may affect uptake dynamics are addressed in this dataset.

For terrestrial receptors, it is assumed that the bioavailability of each ECOPC in Site media is similar to the toxicity tests used to derive the TRV. Many toxicity tests are performed using soluble forms of chemicals so that dose can be effectively controlled in the test. The chemical forms found in the environment for many chemicals, especially metals, is lower than assumed for toxicity tests. It is possible, although less likely, for the chemical forms at the Site to be more bioavailable than the forms used in toxicity testing. As a result, the exposure estimations for all terrestrial and riparian receptors are expected to be overestimated to an unknown degree but the possibility exists that exposure may be underestimated to an unknown degree.

5.5.3 Toxicity and TRVs

In general, risk assessments draw from information gained from laboratory and other carefully controlled experimental exposures, which is then used to extrapolate conditions likely to exist in the natural environment. The laboratory information often does not provide complete linkages for these extrapolations. Consequently, assessment factors are often used to compensate for the many uncertainties inherent in the extrapolation from laboratory effects data to effects in natural ecosystems (Warren-Hicks and Moore 1998). As described in Calabrese and Baldwin (1993), uncertainties arise when extrapolations are made from the following:

- Acute to chronic endpoints;
- One life stage to an entire life cycle;

- Individual effects to effects at the population level or higher;
- One species to many species;
- Laboratory to field conditions;
- One to all exposure routes;
- Direct to indirect effects;
- One ecosystem to all ecosystems; and/or
- One location or time to others.

The net effect of these uncertainties may result in either an overestimate or underestimate of risk, depending on site-specific conditions, the types of receptors included in the evaluation, and the ECOPCs. To limit this uncertainty to the extent possible, chronic toxicity endpoints indicative of reproductive, growth, or mortality effects were used. In addition, for wildlife receptors a range of TRVs was discussed where appropriate.

Care has been taken in the gathering of toxicity values for the ECOPCs at the Site to minimize the points of uncertainty presented in the list above by using well established TRVs that are based on high quality investigations. However, no procedures for the identification of toxicity data can eliminate all or even most of the uncertainty inherent in the process of toxicity assessment.

Selenium in the aquatic system is a particularly important aspect of the SSERA and has been the subject of intensive water quality and ecotoxicity investigations at the Site. The Site-specific studies together with USEPA's recent re-evaluation of selenium aquatic toxicity for the Draft National Selenium Criterion (USEPA 2015), and the host of literature and studies used to derive the Draft National Criterion have reduced uncertainty and establish the following:

- Chronic selenium effects are manifested through dietary uptake of the maternal parent, not aqueous or sediment exposures, and the effects endpoints are either malformations in young developing organisms or survival.
- Egg/ovary tissue concentrations of selenium provide the best relationship to effects in young.
- There is a difference in organism responses based on their environment (e.g., lentic versus lotic).
- The EC_{10} is a sensitive measure of effects.

- Fish are an appropriately sensitive organism to assess effects in aquatic systems.
- Some fish species are more sensitive than other fish species.
- Egg/ovary effects thresholds can be translated to whole body tissue concentrations.

On a site-specific basis, brown trout is one of the more sensitive species tested to date and the most sensitive salmonid species tested. Use of the brown trout tissue threshold (EC_{10}) for this Site reduces many of the uncertainty issues that may arise.

TRVs – Surface Water

The benchmarks or TRVs for a single environmental medium may be derived differently and reflect varying levels of conservatism. For surface water, Idaho state standards or AWQC were the primary TRVs utilized. When neither was available, Michigan Rule 57 Tier 2 values were utilized. Tier 2 values are typically more conservative because they have added safety factors due to use of fewer studies to derive the criterion value. In other cases, such as aluminum, the AWQC have not been updated since 1988 at the national level. For this SSERA, a Region 6 approved aluminum criteria set was used that included a hardness-based derivation that more accurately reflected toxicity of aluminum in surface waters in the pH range from 6 to 9. For two metals, manganese and uranium, CDPHE standards developed for the Arkansas River basin were used to assess effects due to these parameters. Both are hardness-based criteria developed using the USEPA approved methods for criteria derivation. Overall, nearly all of the surface water criteria were derived using a similar process, prescribed by USEPA, to arrive at a chronic value. While the process allows for different levels of conservatism (i.e., standards and AWQC versus Tier 2 criteria), it is a methodical derivation process.

However, very limited data were available for iron. While USEPA (1986) is cited, the iron value has its origins back in the USEPA Red Book circa 1976. A single field effect study was used as the basis for the 1 mg/L value, and there has been little corroborating evidence since. For this reason, EPRI (2004) undertook the task to develop a scientifically defensible iron criterion; however, they found that the availability of usable laboratory studies was limited. Instead they used an extensive bioassessment dataset from West Virginia to derive a value consistent with the CWA 101(a) goal (i.e., maintain the chemical, physical, and biological integrity of the Nation's waters). The result of their analysis was that a value of 1.74 mg/L would be protective of those aquatic biological communities. While it is not intended that this value is applicable to the Site, the research from that document and compilation of studies reviewed indicates that a number of factors govern iron toxicity, which is likely site-specific due to these compounding factors. Therefore, the surface water TRV for iron may have significant uncertainty and is likely biased low for this Site.

TRVs – Sediment

Some sediment benchmarks will use a 15th percentile value for a low or no effect benchmark and an 85th percentile for high effects. Others will be even more conservative and only consider a concentration to be protective if at a 95 percent confidence level (i.e., individual level protection versus population level protection). These threshold levels are expressed as effects range-low (ER-L), apparent effects threshold (AET), or upper effects thresholds (UET), to name a few. Collectively, these values and others like them are often referred to as SQGs. Because SQGs do not exist as national regulatory thresholds and for the most part are still in a state of development on regional or state levels, there may be varying levels of effect threshold values for a single parameter.

MacDonald et al. (2000) indicate that the numerical SQGs for any particular substance can differ by several orders of magnitude, depending on the derivation procedure and intended use. Despite the wide range of benchmark values, derivations procedures, endpoints, etc., there has been an ongoing effort in the last 5 to 8 years to develop SQGs that are consensus-based values. The apparent goal of such efforts is to integrate the various studies conducted to date to develop a two-tiered benchmark that is backed by real data from many field and laboratory studies which results in benchmarks that are broad-based, more accurate as to the level of toxicity, and correlated to effects. For these reasons, the TEC and PEC values from MacDonald et al. (2000; 2009) were used to screen sediments and assess potential risks. Essentially, these are the best available thresholds based on the most rigorous evaluation of matching concentrations in sediments to toxic/not toxic endpoints. When TEC/PEC values were not available, ER-L, ER-M, or AET, UET type thresholds were used, if available.

Despite the availability of SQGs for some parameters, not all have had these types of values developed. For example, no SQGs were found for beryllium, boron, molybdenum, thallium, and vanadium. Literature-derived toxicity values were found for all ECOPCs without SQGs except for beryllium and vanadium. For barium, the screening value used has no basis in aquatic toxicity, but rather a dredging guideline for sediments. TRVs for no-effect and low-effect exposures were identified for benthic tissue from literature sources to further estimate potential selenium risks. These TRVs were based on dietary uptake to mayflies, which are generally considered among the taxa that are more sensitive to environmental toxins, especially metals.

One important uncertainty related to sediment TRVs and assessment of sediment risks is related to the definition of risk or no risk for those sediment concentrations that fall between the TRV_{low} and TRV_{high}. This is a gray area between no or very low levels of effects and likely levels of effects. There is no clear definition to suggest that, at some threshold in this gray area, effects are likely. Several considerations also play into this assessment such as exposure through a complete pathway, duration of exposure, and magnitude of exposure.

TRVs – Fish Tissue

A variety of different fish tissue thresholds from varying studies reported in the literature were utilized. The hierarchy for selection of the fish tissue TRVs was previously described. Uncertainty in the use of these tissue thresholds arises from several sources, many of which have been discussed above, including bioavailability of the contaminant, exposure route, test duration, species used for testing, and/or the endpoints measured, among others. Of the ECOPCs assessed in this SSERA, all ECOPCs have toxicity thresholds established due to exposure to surface water concentrations. For most of these ECOPCs, water exposure would be the primary medium by which risk would be predicted. While representative and conservative tissue TRVs were compiled for trout and other species of fish, the first line of evidence for risk prediction should be considered the surface water concentrations.

Selenium, while also including a surface water effects threshold, is undergoing a fundamental change in the basis of how toxic effects will be assessed due to a better understanding of how selenium affects fish. The tissue effect threshold for selenium used in the SSERA is based on developmental and survival effects in brown trout. USEPA included Simplot's brown trout adult reproduction studies (Formation 2011c, 2012c; and AECOM 2012) in its derivation of the Draft National Selenium Criterion (USEPA 2015). See Appendix D for a more detailed discussion of the derivation of the brown trout tissue thresholds, both by USEPA and by Simplot.

Examination of the Site population data also suggests that the upper effects threshold selected for use as a fish tissue TRV may also be conservative. This is discussed in more detail in the following section.

5.6 Population –Level Risk Characterization

The most apparent uncertainty associated with the assessment of risk is the extrapolation of assumptions about the potential for adverse effects from individual organisms to populations or communities. Few models are available to extrapolate the potential for adverse effects from the individual level to the population or community level. Because of the limited availability of such models, certain assumptions had to be made about the overall potential for adverse effects to ecological receptors. It is generally assumed if there is no potential for direct adverse effects to individual organisms, then the potential for direct adverse effects to populations or communities is unlikely. For this reason, both NOAEL and LOAEL based TRVs are used in the analysis of risk. It is generally assumed that if there is no risk to a receptor predicted using a NOAEL TRV, then no risk is predicted on an individual level which can be assumed to be protective at the population level. It is also generally assumed that if exposure to a receptor is predicted to be lower than the LOAEL TRV, then no risk is predicted at a population level.

Empirical data for populations can also be used to reduce the uncertainty of extrapolating from individual effects to population-level effects provided sufficient spatial and temporal information is available. For this Site, trout population data have been collected annually (from 2006 to 2015) at a subset of locations each fall that represent a range of selenium exposures. While population variability is present from year to year due to changes in flows, temperatures, and other habitat factors, Site-specific population monitoring data provide the best evaluation of potential changes that may occur in response to contaminant exposure.

5.7 Uncertainties Related to Background Influence

The background concentration of ECOPCs may factor into risk estimates, particularly where widespread risks are predicted without obvious sources or in known unimpacted areas. Naturally elevated background concentrations of inorganics are common in mineralized zones. Background considerations may be incorporated into the assessment and investigation of sites, as acknowledged in existing USEPA Guidance (USEPA 2002a, 2002b). As discussed in the SSERA Work Plan, no additional data collection to determine background concentrations of ECOPCs was planned. However, specific sampling and analysis activities needed to characterize background may be developed if needed.

5.8 Uncertainties Related to the Hazard Quotient Approach

It is generally accepted that HQs greater than 1.0 may indicate the potential for risk to an ecological receptor. However, there is no consensus regarding the degree of divergence from 1.0 that indicates a potential for risk to receptors. Menzie et al. (1992) point out, however, that an HQ greater than 1 by itself does not indicate the magnitude of effect nor provide a measure of potential population-level effects.

The SSERA uses both the NOAEL and LOAEL TRVs for wildlife to present a range of potential risks for each receptor. By providing a range of HQs, the uncertainty associated with the HQ approach is decreased.

5.9 ECOPCs without Toxicity Data

The SSERA screening process identified a list of ECOPCs that were detected at the Site but did not have adequate TRVs available for either avian receptors or for plants and invertebrates. Risks for these ECOPCs are uncertain, but low relative to the remaining ECOPCs discussed in Section 4.3.4.

6.0 CONCLUSIONS

A tiered assessment approach was used to estimate risks of ECOPC concentrations to aquatic and terrestrial receptors at different potential habitats and exposure scales at the Site. Tier 1 assessment reflects the traditional approach of calculating Site-wide exposure point concentrations. Tier 2 estimates exposure separately for the mine panels and reclamation types, and is intended to help evaluate the relative effects of the varying reclamation approaches on potential risk. Tier 3 is intended to identify the parts of the Site that contribute most to overall exposure and risk estimates. The location-specific HQs in Tier 3 are shown as indicators of relative exposure, but do not represent independent estimates of risk for assessment endpoints because they do not reflect ecologically meaningful areal scales.

Selenium is the primary risk driver for both aquatic and terrestrial habitat. While exposure to other ECOPCs exceeded risk benchmarks in some areas, the elevated concentrations coincided with elevated selenium exposures in most cases. Conclusions for aquatic receptors are presented by media type to reflect the risk analysis organization and regulatory framework for aquatic environments. Terrestrial risk analysis is based on ingestion of ECOPCs from multiple exposure media within each habitat. Terrestrial risk conclusions are presented by habitat type and ECOPC. A summary of the risk characterization and conclusions follows.

6.1 Aquatic Resources

The three tiers of risk characterization demonstrate that selenium is the primary risk driver in surface waters across several drainages. Exposures of aquatic receptors to other ECOPCs in primarily surface waters including aluminum, arsenic, cadmium, iron, nickel and zinc also exceeded TRVs (almost always at Pole Canyon Creek), but do not likely represent unacceptable risk because of the very restricted exposure (e.g., seeps or ephemeral habitats) of receptors to these environments.

Selenium is the primary ECOPC driving risk estimates and the potential need for risk management decisions. Locations where elevated selenium concentrations exist and pose risk to aquatic receptors correspond to areas of known inputs such as Hoopes Spring and downstream receiving waters and Pole Canyon Creek.

<u>Surface Water</u>

- <u>Selenium</u> – Selenium concentrations in surface water from Pole Canyon Creek ($HQ_{chronic}$ = 1166), North Fork Sage Creek ($HQ_{chronic}$ = 2), Hoopes Spring ($HQ_{chronic}$ = 9), South Fork Sage Creek ($HQ_{chronic}$ = 3), Lower Sage Creek ($HQ_{chronic}$ = 5), and Crow Creek ($HQ_{chronic}$ = 2) exceed the current state standard (0.005 mg/L) for aquatic life (Table 6-1). Using the acute state standard (0.02 mg/L), potentially acutely toxic concentrations of selenium are also found at Pole Canyon Creek (HQ_{acute} = 292) and Hoopes Spring (HQ_{acute} = 2) (Table 3-22). However, as discussed in Section 3.2.1.3, the current state of the science indicates that selenium concentrations in egg/ovary tissues provide the best measure of effects in fish, which are sensitive aquatic receptors. As a result, site-specific egg/ovary and whole body fish tissue data should be the primary media used to assess risk to aquatic receptors for the Site (see Fish Tissues section below). According to USEPA (2015), the fish tissue data should be the primary metric used to evaluate risk to fish when tissue data are available at a location.

- Arsenic, Cadmium, Nickel, and Zinc – Arsenic ($HQ_{chronic}$ = 2), cadmium ($HQ_{chronic}$ = 33), nickel ($HQ_{chronic}$ = 5), and zinc ($HQ_{chronic}$ = 6) concentrations exceed chronic water quality TRVs at the LP-1 location, which is a seep at the downgradient toe of the Pole Canyon ODA (Table 6-1). Cadmium (HQ_{acute} = 13) and zinc (HQ_{acute} = 6) concentrations exceed the acute water quality TRVs at the LP-1 location. However, exposure for aquatic receptors at this location is limited due to the isolated nature of this seep and its lack of connectivity to the lower Pole Canyon Creek drainage, except during years with the largest amount of spring runoff. This connectivity, while ephemeral, may pose a risk to aquatic receptors, albeit the risk is short term and very localized to a small section of North Fork Sage Creek (HQchronic = 2).

- Iron – Elevated iron concentrations in North Fork Sage Creek surface waters (NSV-2 [a spring] and NSV-6) ($HQ_{chronic}$ = 2 for maximum concentration from these two locations) may pose a risk to aquatic receptors, but in a very limited area as elevated iron is not observed downstream. Elevated iron concentrations at the SV-1 irrigation ditch location ($HQ_{chronic}$ = 2) pose a low risk to receptors due to lack of habitat and exposure. In South Fork Sage Creek ($HQ_{chronic}$ = 1), elevated iron concentrations were observed during a high-flow monitoring event, but not observed during the subsequent low-flow monitoring event. Increased iron concentrations are likely episodic with a short-term duration.

Sediment

- Selenium – Selenium in sediments from Hoopes Spring (HS-3) and North Fork Sage Creek (at NSV-6), and at Pole Canyon Creek (LP-PD and LPT-1, LPT-2, and LPT-3 sites) exceeded the sediment TRV. However, the TRVs for selenium in sediments are not based on toxicity to benthic invertebrates. Literature-derived tissue TRVs for benthic invertebrates, compared to concentrations measured for invertebrate tissues collected from across the Site, indicate selenium in invertebrate tissues potentially poses a risk only in lower Sage Creek (NOEC HQs = 2 and 3, while LOEC HQs = 2 for each site).

- Barium – Barium in sediments is elevated upstream and downstream of mine areas as HQs across the Site ranged from 3 to 11 using the TRV_{low} value and 2 to 4 using the

TRV_{high} value (Table 6-1). Potential risks due to barium concentrations in sediments are unlikely due to its low solubility and low bioavailability.

- Cadmium – Cadmium in sediments was elevated in Pole Canyon Creek (HQ_{high} ranged from 4 to 9) and an irrigation ditch near Sage Creek (SV-1) (HQ_{high} = 3). Elevated cadmium concentrations in sediments in lower Pole Canyon Creek and at sampling location SV-1 were only present in areas where flow is ephemeral at best and no appreciable aquatic habitat is present. Therefore, risk to aquatic populations is low due to an incomplete pathway for exposure.

- Chromium, Nickel, Silver, and Zinc – Chromium, nickel, and zinc in lower Pole Canyon Creek sediments exceeded their respective TRV_{high} for these ECOPCs resulting in HQs > 1. Silver only exceeded the $TRV_{low.}$ Similar to cadmium, the pathway for exposure is incomplete. Thus, risks due to these ECOPCs are expected to be low.

- Manganese – North Fork Sage Creek was the only drainage with a manganese concentration that exceeded the TRV_{high} value (HQ = 5) (Table 6-1). Smoky Creek, Tygee Creek, Pole Canyon Creek, Sage Ceek, and South Fork Sage Creek all had sediment concentrations that resulted in HQ_{low} greater than 1.

- Metal Mixtures – Evaluation of potential cumulative risks due to metal mixtures in sediments using the mean PEC–Q[5] identified Pole Canyon Creek drainage and Sage Creek drainage sediments as potentially posing a risk to aquatic benthic macroinvertebrates due to concentrations of multiple ECOPCs. Sage Creek was identified but it was clearly a function of a single site (SV-1) where consistently higher metal concentrations were found. In Sage Creek proper, sediment metals concentrations were consistent low. However, as mentioned above, the pathway for exposure is incomplete, as connectivity to downstream waterbodies is limited and inconsistent.

Fish Tissue

- Selenium – Selenium in fish tissue is the best measure of exposure and potential risk for fish and other aquatic receptors (USEPA 2015). Tissue based effects thresholds (EC_{10}) were developed using Site-specific studies on brown trout and YCT (Formation 2012c). Of these two species, brown trout was found to be more sensitive to the effects of selenium. Subsequent evaluations and reviews of the brown trout data through numerous peer reviews (USEPA 2015, ERG 2012, GLEC 2014) have yielded a range of potential effects thresholds in egg/ovary tissues from the brown trout study using endpoints based on survival, deformities, and a combined endpoint of survival and deformities.

[5] The PEC-Q approach is a valid approach for metals that have consensus-based guidelines. No consensus-based guideline has been developed for selenium in sediments.

Whole body selenium tissue concentrations downstream of major sources exceeded the USEPA-derived and Simplot-derived tissue thresholds at Hoopes Spring (HQ = 2) and Lower Sage Creek (HQ = 2). Overall, fish tissue from Crow Creek and South Fork Sage Creek do not currently exceed the USEPA-derived or Simplot-derived site-specific tissue thresholds, although some fish tissue samples have selenium concentrations that are approximately equal to the site-specific tissue threshold. Data on brown trout populations (about 8 years of standing crop biomass data) suggest that trout subpopulations inhabiting areas just downstream from sources may be adversely affected when whole body tissue concentrations exceed about 24 mg/kg dw. A noted decline in population biomass was observed at whole body tissue selenium concentrations greater than 24 mg/kg dw in whole body tissue, a value higher than the USEPA-derived and site-specific tissue thresholds derived for the Site (12.48 and 14.14 mg/kg dw whole body, respectively). Average concentrations have exceeded this level only at HS-3, LSV-2C, and LSV-4, but not in more downstream areas such as Crow Creek. If selenium concentrations from sources increase in areas such as South Fork Sage Creek or Crow Creek downstream of Sage Creek, risks to populations may become significant.

- Aluminum, Copper, Iron, and Zinc – Aluminum, copper, iron, and zinc in fish tissues were elevated at several, if not all, Site locations evaluated. Of these, copper, iron, and zinc are essential micronutrients. The contributions of background to tissue concentrations, as well as the reliability of the TRVs used to assess potential risks, are discussed in the Uncertainty Analysis (Section 5).

Concentrations of ECOPCs in biotic and abiotic media exceeded TRVs for aquatic receptors. However, whether these concentrations represent significant ecological risk is often a function of habitat and connectivity of surface water among sites. Distribution through the various streams is based on the level (or lack) of connectivity to these source areas. The primary sources of selenium in surface waters downstream from the Site (e.g., Hoopes Spring and South Fork Sage Creek springs) are currently being treated under a treatability pilot study designed to reduce selenium concentrations in the groundwater where it discharges to surface water. A trend of decreasing selenium concentrations with distance from the source areas is evident (without treatment), thus control and reduction of selenium concentrations at the two primary source areas will have concomitant reductions in Sage Creek and Crow Creek.

Similarly, at Pole Canyon Creek, the diversion of upper Pole Canyon Creek water around the ODA has proven to significantly reduce selenium in surface waters downstream of the ODA; surface water that formerly flowed in contact with the base of the ODA now bypasses the ODA. However, the ODA east toe seep (LP-1) discharges a small volume of water, derived from infiltration of snowmelt and stormwater through the ODA materials, at the toe of the ODA resulting in significant metals concentrations, including selenium, cadmium, nickel, and zinc. However, because upper Pole Canyon Creek is diverted around the ODA, the LP-1 seep provides limited exposure potential to these high concentration waters. Source management, including construction of a 5-foot thick cover in 2015 under a Non-Time-Critical Removal Action, should continue to prove effective in reducing potential surface water risks at the Site.

6.2 Terrestrial Resources

Selenium is the primary ECOC driving risk estimates and the potential need for risk management decisions at the Site. Other ECOCs were identified as a result of Tier 1/Tier 2 analysis including cadmium, copper, lead, vanadium, and zinc identified as ECOCs for both the upland and riparian receptors. Chromium, manganese, and molybdenum were identified as ECOCs for the riparian receptors only.

Elevated concentrations of ECOCs were observed primarily in mined areas with either no cover (i.e., direct revegetation of overburden) or topsoil-only reclamation. Elevated concentrations of ECOCs in soils corresponded with higher exposure and risks. Exposure and risks were considerably lower for northern Sage Valley, Panel A Area 1, and Panel E (Figure 6-1). Risks were lowest within the Dinwoody reclamation type and highest in the areas with no cover.

For areas of northern Sage Valley that are outside of the Pole Canyon Creek corridor, concentrations of ECOCs are low and may reflect background concentrations. However, identification and approval of Site-specific background concentrations may be necessary to support this conclusion. The Tier 3 risk characterization conclusions are summarized in Tables 6-2 (upland receptors) and 6-3 (riparian receptors) for all of the ECOCs.

<u>Selenium Exposure and Reclamation Status</u>

- Overall, risks were highest in those panels shown in Table 4-31 as high exposure panels. Those panels and northern Sage Valley identified as low exposure panels have considerably lower risk from selenium exposure. Selenium exposure and risk within Panel A Area 2, Pole Canyon ODA, Panel D North, and Panel D South is higher than the areas discussed in the previous bullet. HQs using the lowest LOAEL TRV ranged from 7.3 for the northern harrier to 56 for the deer mouse in A Panel Area 2. On the Pole Canyon ODA, HQs ranged from 6.2 for the northern bobwhite to 38 for the deer mouse. On Panel D, HQs ranged from 10 for the northern bobwhite to 64 for the deer mouse in the northern section and from 7.9 to 49 for the same receptors in the southern section.

- Exposure is highest in Panel A Area 2, Panel D North and South, and the Pole Canyon ODA (Table 4-31) which represent areas where exposure to selenium-bearing overburden materials is expected to be highest.

- The Dinwoody mine reclamation type (Panel E Area 1) has been successful in reducing selenium exposure in the terrestrial foodweb, thereby, significantly reducing risk compared to areas covered by topsoil only or where no cover was placed. Selenium exposure in the Panel E Area 1 Dinwoody reclamation was similar to the exposure predicted for relatively unaffected areas in northern Sage Valley outside of the direct influence from the mine.

HQs calculated using the lowest-LOAEL TRV in the Dinwoody Cover areas ranged from less than 1 for the coyote, eastern cottontail, mule deer, northern bobwhite and northern harrier to 1.9 for the American robin and 4.5 for the deer mouse. Low HQs relative to the rest of the mine panels were also observed within northern Sage Valley, ranging from 2.1 for the Northern Harrier to 14 for the Deer Mouse.

While HQs greater than 1.0 were calculated in the Dinwoody reclamation areas and northern Sage Valley, there is a lack of significant pathways to mining-related exposure in those areas. As a result, the HQs in the Dinwoody areas and in northern Sage Valley represent areas that have little or no chemical impact from mining and should be considered at a baseline level of risk. .

- Estimated exposure to selenium is predicted to be similar to the areas reclaimed with Dinwoody material within Panel E Area 2 and Panel A Area 1, both reclaimed using topsoil cover over chert layers. HQs based on the lowest LOAEL TRVs in those areas were similar to those observed in affected parts of northern Sage Valley ranging from 2.2 for the northern bobwhite to 9.5 for the deer mouse in Panel A Area 1 and from 2.0 for the northern harrier to 11 for the deer mouse in Panel E Area 2.

- For selenium, the NOAEL TRV and the lowest LOAEL TRVs in the EcoSSL database are very similar and do not provide risk managers with a good estimate of the potential range of exposure at which effects may be observed. This is especially evident for mammalian receptors that have nearly identical NOAEL (0.143 mg/kg BW/day) and lowest LOAEL (0.145 mg/kg BW/day) TRVs. While this may be indicative of a very robust toxicological dataset, nearly identical NOAEL and LOAEL TRVs provide risk managers with only a limited view of the potential for risk.

  In order to provide more useful information on the range of potential risks, the Tier 3 assessment also utilized the geometric mean NOAEL TRV presented in the EcoSSL document as a comparison point. These values provide an estimate of the mean exposure rate across all of the sublethal growth and reproduction endpoints in the database.

  As noted for COCs in Section 4.3, the geometric mean NOAEL from the EcoSSL compilation may be an appropriate TRV for population-level assessment endpoints, instead of the lowest bounded LOAEL, because it reflects levels at which sensitive individuals in a population may be affected, but overall population effects seem unlikely. The geometric mean NOAEL is still very conservative, and other factors including habitat and the extent of local populations are much bigger factors in determining whether ecologically meaningful adverse impacts on a population are expected.

- For the bird receptors, the geometric mean NOAEL TRV was approximately two times greater than the lowest LOAEL TRV. This means that exposure estimates resulting in HQs approximately equal to 2.0 using the lowest LOAEL TRV would be approximately equal to 1.0 using the geometric mean NOAEL TRV. For the mammalian receptors, the geometric mean NOAEL TRV is approximately three times greater than the lowest LOAEL TRV. These results highlight the variability in the toxicity data available for selenium exposure and should be considered accordingly as part of the risk management decisions for the Site. While exposure to the terrestrial receptors and potential risk to individual receptors and species is elevated compared to the surrounding areas, it is unknown whether any actual effects are occurring to the populations inhabiting those areas. No data are currently available to address the actual presence or absence of population-level effects from selenium as predicted in this assessment.

Selenium Exposure and Riparian Habitats

- As for the more upland terrestrial receptors, elevated selenium in the semi-aquatic habitats at the Site was limited to a few sampling locations. At seeps DS-7 (east of Panel D) and ES-4 (east of Panel E) as well as riparian location LP-PD (Pole Canyon), selenium exposures were much higher than elsewhere. At seep DS-7, HQs calculated using the lowest LOAEL TRV ranged from 5.4 for the raccoon receptor to 83 for the deer mouse receptor. The calculated HQs at LP-PD were similar, ranging from 1.4 for the mink receptor to 88.1 mg/kg for the meadow vole. HQs were highest at ES-4, ranging from 10 for the raccoon to 320 for the meadow vole receptor.

- Risk was lowest at seep ES-3 (east of Panel E) with HQs ranging from less than 1 for most receptors to 1.4 for the deer mouse and at riparian locations LS (Lower Sage Creek) and LSm (Lower Smoky Creek). At both of those locations, most receptors had HQs less than 1.0 with maximum HQs less than 2.0 for the most highly exposed receptors.

- Risk at Hoopes Springs and LSS (Lower South Fork Sage Creek) was intermediate. At Hoopes Springs, most receptors had HQs less than 5 and at LSS, all HQs were equal to 5 or less.

Copper in Small Mammals

- Highly elevated concentrations of copper detected in several of small mammal samples were observed and resulted in exposure estimates that exceeded the TRVs for both the coyote and northern harrier receptors. The concentrations observed were elevated above any concentrations of copper expected at the Site and showed no clear spatial relationship or relationship with copper in surface soils on the reclaimed areas of the mine.

- The HQs calculated using the lowest LOAEL were highest at the Pole Canyon ODA (HQ = 18 for the coyote and 85 for the northern harrier). HQs greater than 10 were also calculated for the northern harrier receptor in Panel A Area 1, Panel D (North and South), and northern Sage Valley.

- Site-wide risk to both the coyote and northern harrier cannot be ruled out. However, due to the apparently anomalous copper data in small mammals in relation to copper measured in surface soils on the reclaimed areas, there is considerable uncertainty in that conclusion and further study is recommended prior to making risk management decisions related to copper exposure for carnivorous receptors.

- A proposed plan for additional sampling of small mammal tissues in the areas where the suspected anomalous copper results were observed will be provided to the Agencies in early 2016. Collection of the samples is expected in the summer of 2016.

Other ECOCs

Concentrations of the following ECOCs correspond to exposures that exceed LOAELs at some locations: cadmium, chromium, lead, manganese, molybdenum, vanadium, and zinc. Exposure of LOAELs for these ECOPCs is restricted to small portions of the Site. Therefore, while individual receptors may experience exposures exceeding LOAELs, overall effects from these chemicals on populations is low. Risk management decisions for wildlife should, therefore, be based on the potential risk from selenium exposure.

Concentrations of other ECOCs in surface soils exceeded one or both of the plant and invertebrate screening levels in Tiers 1 and 2 of the risk characterization. Plant communities, as well as ECOC concentrations in plants and invertebrates are similar for northern Sage Valley and Dinwoody reclamation areas in the ODAs. The presence/absence of plant species and community composition are likely to be affected by physical factors such as soil texture and nutrient availability, especially in overburden areas. Therefore, communities in un-reclaimed ODA areas and the active mine area may be affected by a combination of physical habitat factors and ECOC concentrations. However, ecological function of population and communities in northern Sage Valley sites are likely unaffected.

No Site-specific data are available on background concentrations of the ECOCs. However, evaluation of data from other sites and guidance from other western states suggests that concentrations of some metals in soils at the Site are within natural background ranges. This is especially true for the area of northern Sage Valley where overburden and disturbance from mine activities is not present. Background concentrations are being evaluated for other phosphate mining sites in southeastern Idaho that may provide information relevant to the Smoky Canyon Mine. However, conclusions regarding background concentrations cannot be established until agreement on Site-specific background has been reached.

Risks to ecological receptors have been identified through the risk assessment process described in this SSERA based on expected exposure scenarios. Results will be used to support the development of remedial alternatives and provide a basis for comparing potential ecological impacts of the alternatives in the forthcoming FS for the Smoky Canyon Mine. The conclusions of this SSERA will be available for the support of risk management decisions regarding which remedy is most appropriate for the Site.

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TABLES

FIGURES

APPENDICES A, B, AND C

PROVIDED ELECTRONICALLY ON CD

APPENDICES D AND E

PROVIDED IN HARD COPY AND ELECTRONICALLY ON CD